Bird Sensitivity Map to provide locational guidance for onshore wind farms in Scotland

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Executive summary

In response to concerns over climate change, the UK Government has set a target of a 60% reduction in carbon dioxide emissions by 2050. It is proposed that this target will be achieved in part by switching energy generation from the burning of fossil fuels to forms of generation with lower carbon dioxide emissions, such as renewables. The UK has a target for 10% of its energy to be generated from renewable sources by 2010, and the equivalent target for Scotland is 20%. In 2002, the UK Government introduced the Renewables Obligation, and associated Renewables Obligation (Scotland), a market support mechanism, which requires that power suppliers must source increasing amounts of their energy from renewables. Wind is currently the cheapest and most technologically advanced form of renewable energy, and thus the Renewables Obligation has led to a huge increase in the number of wind farm proposals. The wind resource, coupled with reluctance to have wind farms close to habitation, has led to many of these sites being in the uplands. Scotland contains a large area of upland habitat and has the highest number of schemes being considered for planning proposals of any of the UK countries. Scotland's upland habitat supports many important populations of birds of conservation concern, and this, combined with the negative effects wind farms can have on birds, leads to potential conflict.

In order to help minimise this conflict, a bird sensitivity map to aid location of onshore wind farms in Scotland has been created, based on distributional data for a suite of sensitive bird species. Species included on the map are either listed on Annex I of the EU Birds Directive, and/or are species of conservation concern with known or suspected susceptibility to the effects of wind turbines on birds, notably collision mortality and disturbance displacement. The sensitivity map has been produced at a 1km square resolution, with each 1km square in Scotland being assigned one of three sensitivity ratings. These sensitivity ratings were assigned following reviews of literature and best available information for each species on foraging ranges, collision risk, disturbance distances and other relevant features of behavioural and population ecology, to develop 'sensitivity criteria' to determine appropriate buffering distances to apply to the distributional data for birds.

The map indicates that there is a greater incidence of bird sensitivities in north-west Scotland. Particularly sensitive areas occur in the Highlands, Western Isles and Northern Isles. Thirty seven percent of the area on the map is classified as 'high sensitivity' (31 418 1km squares), thirty one percent as 'medium' (26 358 1km squares) and thirty two percent as 'low/unknown sensitivity' (26 813 1km squares).

The following caveats need to be taken into account when using the map:

- The map has been developed from the best information available. However, data deficiency means that the map cannot be comprehensive. Gaps in survey coverage for species on the map mean that there is no guarantee these species will not occur in 'low/unknown' sensitivity squares. Lack of available data also means that some sensitive species could not be included on the map. No liability is accepted for the presence or absence of species at particular sites contrary to that indicated on the map.

- The map is based primarily on the locations of breeding birds.

- The sensitivity map was created by collating data that were collected for other purposes, and thus not tailored specifically to the map's requirements (which would be an enormous task).

- The map is not a substitute for Environmental Impact Assessment, but is intended as an indicative map of likely bird sensitivities, to help guide decision-makers in the early stages of the planning process.

- The map will require updates to add new survey data for the species included, to add data for further species as they become available, and to reflect revisions to the sensitivity criteria in the light of new information, notably from research and experience at working wind farms.

Contents

Introduction

Climate change poses the single biggest long-term threat to birds and other wildlife, with mid-range climate warming scenarios predicting that 15% to 37% of species world-wide will be ‘committed to extinction’ by 2050 (Thomas *et al*., 2004). Annual temperatures across the UK are predicted to rise by between 2°C and 3.5°C by the 2080s (Hulme *et al*., 2002). In 2001, the IPCC predicted that the globally averaged surface temperature will rise by 1.4°C to 5.8°C between 1990 and 2100, but more recent analysis suggests the upper limit of possible warming could be as much as 11°C (Stainforth *et al*., 2005). These increases in temperature are likely to be accompanied by other changes, such as sea-level rise due to melting ice sheets, and an increase in the frequency or intensity of extreme weather events (e.g. increase in frequency of droughts and flooding caused by El Niño; increased precipitation intensity; increase in tropical cyclone wind intensities (IPCC, 2001)).

There is evidence that climate change is already adversely affecting some species (Crick *et al*., 1997, Parmesan *et al*., 1999, Parmesan & Yohe, 2003, Root *et al*., 2003). For example, the large scale breeding failure of seabirds in 2004 on the North Sea Coast of Britain was thought to have been caused primarily by climate change, leading to a reduction in fish prey (Eaton *et al*., 2005, Mavor *et al*., 2005). Warmer spring temperatures have led to a mismatch in the first laying dates of Golden Plover *Pluvialis apricaria* and the emergence of their tipulid prey, which could have a significant impact on breeding success (Pearce-Higgins *et al*., 2005), and a link between warmer, drier summers and declines in Ring Ouzel Turdus torquatus populations has been suggested (Beale *et al.*, 2006). Declines of around 90% over the past two decades in some populations of Pied Flycatchers *Ficedula hypoleuca* in the Netherlands were also associated with mismatches in timing of breeding and peak food abundance attributed to climate change (Both *et al*., 2006). There is strong suspicion that at least one species has already been driven to global extinction by climate change (the Golden Toad *Bufo periglenes*, a species endemic to Costa Rica (Pounds *et al*., 1999, Still *et al*., 1999)).

In an attempt to minimise the effects of climate change, the UK government has set a target to reduce carbon dioxide emissions by 60% by 2050 (DTI, 2003). There are two main ways of achieving this target; by reducing energy demand, and by switching energy generation from burning of fossil fuels to forms of generation with lower carbon dioxide emissions, such as renewable energy. With this in mind, targets have been set to increase the amount of energy coming from renewable energy sources with a proposed 10% of the UK’s (DTI, 2003) and 20% of Scotland’s (Scottish Executive, 2001) energy coming from renewable sources by 2010. At the end of 2004, 3.1% of the UK’s energy came from renewable sources (DTI, 2006). The Renewables Obligation, and the Renewables Obligation (Scotland) were introduced in 2002 as market support mechanisms to help meet these targets. They require power suppliers to source a specified amount of their electricity from renewables, starting at 3% in 2003 and gradually increasing to 10% by 2010 (BWEA, 2006a). Onshore wind is currently one of the cheapest and most technologically advanced forms of renewable energy, and thus these Obligations have led to an increase in the number of wind farm proposals to meet the increased demand. Their wind resource, coupled with reluctance to have wind farms close to habitation, has led to many wind farm proposals being in the uplands. Scotland contains a large area of upland habitat, and also one of the best wind resources in Western Europe, and thus has the highest number of wind farms being considered for planning approval of any of the countries in the UK, with 43% of schemes under consideration being in Scotland (BWEA, 2006b).

Scotland’s upland habitat supports many bird species of conservation importance, many of which are listed on Annex 1 of the EU Birds Directive (79/409/EEC). Whilst some of these species are concentrated in Special Protection Areas (SPAs), most are more widely dispersed. For example, only

15% of the UK breeding population of Golden Eagle *Aquila chrysaetos* occurs within SPAs (Stroud *et al*., 2001).

Wind farms may have deleterious effects on birds in four main ways; collision, displacement due to disturbance, barrier effects and habitat loss (for review see Langston & Pullan, 2003, Drewitt & Langston, 2006), although the first two are of most relevance to the mapping exercise. Effects tend to be site and species-specific, and the literature on these effects is still limited, with there being few published, peer-reviewed scientific papers. Many studies lack before/after, or wind farm area/reference area comparisons, ignore relevant factors such as assessment of collision risk and differences in bird behaviour between night and day, or are too carried out over too short a time scale to provide conclusive results. There are some robust studies, but many uncertainties remain as to the impacts of wind farms on birds, particularly with respect to the cumulative impacts of multiple wind farms. The BACI (Before-After Control Impact) approach, where data are collected before and after construction, on the assessment area (i.e. the (proposed) wind farm area) and reference areas is the ideal form of study (Anderson *et al*., 1999). Guidelines such as those from SNH (2005) are intended to improve the standard of Environmental Impact Assessments and to promote gathering of more comparable information.

Careful location of wind farms, with respect to sensitive bird species, is crucial to minimise or avoid deleterious effects (Langston & Pullan, 2003). Due to the dispersed nature of many Annex 1 species as mentioned above, it is not sufficient to rely on the protected sites network alone to indicate areas where bird/wind farm conflicts are likely to arise. Furthermore, not all bird species are equally affected by wind farms.

A study for the Scottish Executive, *Scotland's Renewable Resource 2001*, identified a theoretical potential of 11.5GW (Gigawatts) of wind energy available at 3p/kWh generation cost in Scotland (Snodin, 2001). This is roughly equivalent to Scotland's energy consumption, and would require less than 2% of Scotland's land area (Snodin, 2001). The figure of 11.5GW as a potential capacity available in Scotland is a conservative one, as it is capped by an upper capacity limit applied to each Local Authority, proportional to the area of the Local Authority, and based on what was thought to be socially acceptable. Thus, the capacity available was based on this 'social acceptability' constraint, with models being run to look at where this capacity could be most economically placed under various different constraint scenarios.

A strategic approach to planning the location of wind farms is of benefit firstly in safe-guarding bird species of conservation concern, many of which are protected under European law. These species are also an important asset to Scotland's tourist industry, which is Scotland's largest employment sector (Dickie *et al*., 2006). For example, visitors to Osprey *Pandion haliaetus* viewing sites bring around £3.5 million per year to surrounding areas, and Mull's White-tailed Eagle *Haliaeetus albicilla* population is worth £1.4-1.6 million per year in tourism revenue (Dickie *et al*., 2006). Secondly, the strategic approach is valuable to facilitate wind farm development in appropriate locations across Scotland, without wasting resources in preparing, assessing, and determining controversial and often ultimately rejected schemes.

Here, we report on the construction of a bird sensitivity map to aid location of onshore wind farms in Scotland, based on the distributions of a suite of sensitive bird species of conservation concern, including the Special Protection Area (SPA) network. These species are listed on Annex I of the EU Birds Directive and/or species with known or suspected susceptibility to the effects of wind turbines, notably collision and disturbance displacement (Drewitt & Langston, 2006, Fox *et al*., 2006). The map has been produced by drawing on the literature and best available information to determine buffer zones and 'sensitivity ratings' to apply to the distributional data.

Methods

The map presents a strategic view of the sensitivities of bird species of conservation concern in Scotland to onshore wind farm development. It provides an indication of where wind farm development is most likely to come into conflict with bird sensitivities. It is not intended to be prescriptive at the level of the individual site, and does not replace the need for the standard site-based Environmental Impact Assessments.

Species list

The sensitivity map is based on the distributions of eighteen bird species (Table 1). Literature on the behaviour and ecology of these species suggests that they are, or may be, sensitive to wind energy developments, and fourteen of the eighteen species included are listed on Annex 1 of the EU Birds Directive. The four non-Annex 1 species included on the map are Bean Goose *Anser fabalis*, Common Scoter *Melanitta nigra*, Black Grouse *Tetrao tetrix*, and Arctic Skua *Stercorarius parasiticus*. Bean Geese were included on the sensitivity map despite not being an Annex 1 species due to the small, localised nature of their Scottish population, which occurs at just one site, and the fact that geese are potentially vulnerable to both disturbance displacement (Kruckenberg & Jaene, 1999) and collision risk (Hötker *et al.*, 2004) from wind farms. Common Scoter were included due to their small, localised breeding distribution, their red-listed status (Gregory *et al.*, 2002), and the fact that they have been found to be particularly sensitive to disturbance displacement offshore, including in relation to wind farms (Guillimette *et al.*, 1998, 1999, Mitschke *et al.*, 2001, Exo *et al.*, 2002, Kaiser, 2004). Whilst it is not clear whether their offshore sensitivity is relevant at onshore breeding locations, this was considered an appropriately precautionary approach to take. Black Grouse were included because of their rapid rate of population decline in the UK (Sim *et al.*, in prep.), low dispersive ability, and susceptibility to disturbance displacement (Anon., 2003, Langston & Pullan, 2003) and collision (Bevanger, 1995, Hötker *et al.*, 2004). They are also a species of high conservation concern in the UK (Gregory *et al.*, 2002) and one of the top 20 priority species listed on SNH's Draft Species Framework (SNH, public consultation). Arctic Skua were included due to their recent population declines, and as they are a colonial breeder with relatively low SPA coverage (24%). There may therefore be important Arctic Skua colonies that are unprotected by the designated site network.

Several sensitive species were not included due either to lack of recent or comprehensive data, or problems with data access. These species include Osprey, Merlin *Falco columbarius* and Short-eared Owl *Asio flammeus*, for which national surveys are scheduled in the next few years, and could be included in future updates to the map (see discussion).

Special Protection Areas

As well as the distributional data for the eighteen sensitive bird species, SPAs were included on the map. SPAs protect the habitat of rare, threatened or migratory bird species under the EU Birds Directive (79/409/EEC). The requirements for protection of these sites within the European Directives and associated regulations are stringent, and proposals should not adversely affect the integrity of these sites. This includes avoiding adverse impacts on the species for which a site is designated, and avoiding deterioration of or damage to any habitats on which they depend (Stroud *et al.*, 2001). It was not considered appropriate to apply a standard buffer to SPAs, because buffers would differ for different species (and individual SPAs may have several of the included species). Also, where buffers were intended to include, for example, geese feeding areas, these would be unlikely to be uniformly distributed around the SPA boundary, and could be a considerable distance from the boundary. However, areas used by the designated species outwith the SPA should be identified during site-specific assessments (see discussion).

The SPA network was used as a surrogate for areas occupied by species of wintering geese and other waterbirds, as well as colonial nesting seabirds (although see discussion for its limitations). These are congregational species for which a high proportion of their populations occur within the SPA network. The coverage as a percentage of the British populations of the wintering geese and other waterbird species for which the SPA network was used as a surrogate is as follows: Whooper Swan 44%; Greenland White-fronted Goose 59%; Greenland Barnacle Goose 63%; Svalbard Barnacle Goose 79%; Icelandic Greylag Goose 57% and Pink-footed Goose 82% (Stroud *et al*., 2001). Similar values for Annex 1 colonial breeding seabirds are: Arctic Tern 38%; Little Tern 67%; Roseate Tern, 88% and Storm Petrel 35% (Stroud *et al*., 2001). Several species of waterbird and seabird have been identified as being at potential risk of disturbance displacement and collision from wind turbines by Langston and Pullan (2003) and Hötker *et al.* (2004).

Data sources

Many species included on the sensitivity map are monitored either annually or, more usually, on a six or twelve-year cycle as part of the SCARABBS (Statutory Conservation Agency/RSPB Annual Breeding Bird Scheme) surveys. Other species, such as Red Kite *Milvus milvus*, Corncrake *Crex crex*, Slavonian Grebe *Podiceps auritus*, Black-throated Diver *Gavia arctica* and Bean Goose have separate monitoring programmes (sometimes in addition to SCARABBS monitoring, e.g. Black-throated Diver, Corncrake). Survey coverage for most of the species included was comprehensive (Table 1). However, for a few species, e.g. Red-throated Diver *Gavia stellata* and Black Grouse, national survey coverage was low (see Table 1), and thus national survey datasets were supplemented with regional data, collected by RSPB or SNH regional offices, local study groups, or submitted to the Rare Breeding Birds Panel or local bird recorders. Coverage of these regional datasets varied, with some being the result of targeted and fairly comprehensive surveys, whilst others were collections of more *ad hoc* regional records. As the distributional data for these species were the result of a collation of these regional data, rather than the presentation of one national survey dataset, it was not possible to estimate the final coverage achieved.

Although the most recent datasets available were used, the age of these varied for different species. For some species, data from surveys spanning a range of years were used if, for example, it was considered the species was likely to use alternate breeding locations in different years. Some of the species included on the map currently have a national survey either underway or pending (see discussion), and thus the map would benefit from regular updates as new information becomes available.

The map is based predominantly on the locations of breeding birds. This is due to the fact that most national surveys are conducted during the breeding season, when birds are more tied to a particular location (although note that some territorial species are resident year-round, whereas others are seasonally present). For six of the eighteen species, the survey units were nest locations or territory centres (Table 2). In the case of species for which nests are difficult to find, other survey units indicative of breeding behaviour, were used (e.g. churring males for Nightjar *Caprimulgus europaeus*, singing males for Corncrake, lochs with pairs present in the breeding season for Black-throated Diver and Slavonian Grebe) (Table 2). For most species, it was not possible to include areas occupied by immature or non-breeding birds, although for Hen Harrier *Circus cyaneus*, data did not distinguish between locations of nests and sightings, and so both were included in the preparation of the map. The survey units used for each species were taken into account when determining buffer zones to be applied to distributional data.

There were few available data on distributions of wintering or migrating birds. However, some data on wintering birds were incorporated. Red Kite roost locations were included due to their traditional nature and importance to a large number of birds from a currently small, reintroduced Scottish population. Ideally, Hen Harrier communal roost sites would also have been included on the map but this was not possible due to access restrictions to these highly sensitive data. Fields used by wintering Bean Geese were also included on the map as this is a very localised population. As noted above, the SPA network was used as a surrogate for other areas occupied by wintering geese species and waterbirds.

Golden Plover and Dunlin (*schinzii* sub-species, which breeds in the UK) are both listed on Annex 1 of the EU Birds Directive. Both are considered sensitive to wind farm development. However, their relatively high population sizes mean that survey coverage for these species is low (e.g. approximately 3.7% of the Scottish range for Golden Plover). The SPA network includes some important sites for breeding waders (for example the Caithness and Sutherland Peatlands SPA). The network contains 74% of the British breeding population of Dunlin, but just 26% of the breeding Golden Plover population (Stroud *et al.*, 2001).

The inclusion of data from the 1988-91 New Atlas of Breeding Birds (Gibbons *et al.*, 1993) was initially ruled out due to its age and large spatial scale, but it was subsequently decided to include 10km squares with high counts of Golden Plover and Dunlin as 'medium sensitivity' in order to indicate areas where extra care should be taken to check for high densities of these breeding waders. This is outlined in more detail in the sensitivity criteria for these species in Appendix 1 (summarised in Table 2). This approach was used, following unsuccessful attempts to model golden plover distribution, based on regional datasets, and to apply data for breeding waders on SSSIs (see Appendix 2 for details).

Buffering of distributional data

Areas that are heavily used by birds for a greater proportion of the time, notably for activities that are more likely to increase risk to susceptible birds (disturbance displacement or collision), are likely to be the most sensitive locations, and this is the premise of the sensitivity map. Sensitivity criteria (see Species' Sensitivity Criteria for details) were developed, based on territory size, foraging or homing ranges (indicating the relative importance and spatial extent of focal areas for the birds), distances at which species are sensitive to disturbance, collision risk, and other relevant features of behavioural and population ecology. The resulting sensitivity criteria formed the basis for deciding appropriate buffers to apply to the distributional data for each species; for example, the distance by which to buffer nest sites (see Table 2 for summary).

Peer-reviewed published literature was used, where possible, to develop sensitivity criteria, with grey literature being used as a secondary source, although this was treated more cautiously. In the cases of species for which there was little published information on which to base sensitivity criteria, relevant species' experts were consulted and professional judgment applied to determine buffers. Any available information on responses to disturbance displacement caused by wind turbines was used in preference to disturbance distances due to other causes. However, specific, and comparable, information on the effects of wind farms on the different bird species was sparse. Sensitivity criteria were circulated amongst relevant species' experts for comments (see acknowledgements). The amount of information available on which to base the criteria varied between species. For example, for Golden Eagle, radio-tracking has been used to create models to describe the percentage of time spent within different distances of the nest (McGrady *et al.*, 1997, 2002), whereas for other species sometimes little or no relevant information on behavioural ecology was available.

For all of the raptor species as well as Nightjar, Corncrake and Chough, sensitivity criteria were based on estimations of foraging ranges, to reflect the areas within which the risk of collision with turbines, or disturbance displacement whilst foraging, was likely to be greatest. For some of these species (Golden Eagle, Nightjar), the foraging range was divided into two concentric ring buffers, the core area defined as 'high sensitivity' and the outer area as 'medium sensitivity' in order to indicate estimated proportional use of the range (and thus risk of collision, or disturbance displacement whilst foraging).

For loch breeding species; the divers, Slavonian Grebe and Common Scoter, it was not thought appropriate to apply a buffer zone to indicate flight routes to and from feeding areas. For these species, when not feeding on the breeding loch (the frequency of which depends upon the species), most flight activity to other lochs or the sea is likely to occur within 'flight corridors', and a set buffer zone would thus include a large area that was used rarely or not at all. In these cases, buffers were chosen to reflect disturbance distances around the nest or breeding loch, and site-specific assessment should identify corridors around the breeding loch with a high density of foraging flights. For the loch breeding species, where there was a lack of information on disturbance from wind farms or other sources on which to base the buffer zone, a standard buffer of 1km was applied. This was chosen as it was often the 'best-estimate' based on the available disturbance literature for these species. Furthermore, 600m is the maximum reliably recorded disturbance displacement distance for onshore wind farms (albeit for wintering geese, Kruckenberg & Jaene, 1999).

Sensitivity to disturbance is usually determined by measuring the behavioural response to human presence (e.g. flushing distances, distance from nest site to nearest house). However, these distances were interpreted with caution when developing sensitivity criteria, as the response of an individual to disturbance depends upon a number of other factors, such as quality of the site being occupied, distance to other suitable sites and investment made in a site (Gill *et al*., 2001). For example, the fact that an individual does not appear to respond to disturbance could indicate that there are no other available sites for it to occupy, rather than a lack of sensitivity to disturbance (Gill *et al*., 2001). Conversely, the fact that individuals flush at considerable distance from a source of disturbance may just indicate that there is other suitable habitat nearby. Season is also a factor, for example breeding birds are likely to have made a high investment in a site, and thus be reluctant to leave it. All of these factors make escape flight distances difficult to interpret, but in the absence of more detailed information, these may be the best available information.

It is often assumed that collision risk and disturbance displacement are mutually exclusive. However, there may be individual differences in behavioural response and susceptibility due to season or status. For example, at a Norwegian wind farm, some White tailed Eagles have so far ceased to breed within the wind farm (disturbance displacement) whilst some of the birds associated with remaining nests within the wind farm area have collided fatally with turbines (NINA & RSPB, unpublished data).

Sensitivity ratings

Areas on the buffered maps for each species were assigned sensitivity ratings, defined as follows (see Table 2 and section below on Species' Sensitivity Criteria for an explanation of what the ratings mean for each species):

1 **'High sensitivity'** with respect to one or more of the species listed in Table 1 or within an SPA. In these areas, there is a greater chance of conflict between wind farms and sensitive species of birds.
2 **'Medium sensitivity'** for one or more of the species listed in Table 1.
3 **'Low/unknown sensitivity':** not classified as sensitive for any of the species on the list, using the data sources outlined in Table 1 and the sensitivity criteria outlined in Table 2 (see below).

Unfortunately, it was not possible to distinguish between low or unknown sensitivity as gaps in data mean it cannot be ruled out that site-specific assessment may reveal hitherto unknown sensitivities. For example, the area could be important for immature or non-breeding birds (including wintering and migrating birds), which generally could not be mapped. However, areas with a sensitivity rating of '3' will include areas of low sensitivity for birds.

Medium sensitivity indicates that there are historical records for the area but that there is either no subsequent information or the species is now absent (Golden Eagle, Hen Harrier, Slavonian Grebe, Golden Plover, Dunlin). Alternatively, the area forms the outer part of a Golden Eagle or Nightjar's main foraging range and thus may be important to that bird, but is outwith the core range. Areas surrounding Peregrine Falcon nest locations and Black Grouse lek sites have also been classified as 'medium sensitivity'. Thus, medium sensitivity alerts the user of the map to possible sensitivities that will depend upon site-specific assessment to ascertain whether there would be an adverse impact. In some cases, these areas will be found to have a high level of sensitivity.

Creation of sensitivity map

The map was created in MapInfo Professional version 6.0 (MapInfo, 2000). Distributional data (Table 1) were plotted in MapInfo to create a separate data layer for each bird species, plus a layer for SPAs. Most locations were given at the 6-figure grid reference resolution, but some were just given at the 4-figure level. A few were available only at the 2-figure (10km square) level, as they were deemed particularly sensitive by the data suppliers. It was not thought helpful to include data at such a large spatial scale (10km square) on the map. For some species, data are missing from particular areas as locations were considered too sensitive by data collectors (e.g. Hen Harrier on South Uist, Golden Eagle on Skye, mainland Chough *Pyrrhocorax pyrrhocorax*), which reduced coverage slightly for these species.

Once sensitivity criteria and the resulting buffers had been applied to data for each species, the individual species' sensitivity maps were converted to 1km resolution by selecting 1km squares **whose centres fell within** the buffers and assigning these the appropriate sensitivity rating. However, where buffers were very small (<1km) and risked omission of adjacent 1km squares valid for inclusion, the selection was extended to include any squares which intersected in any way with the buffer. Therefore, if the object being buffered was either a 6-figure grid reference (100m square) or an irregularly shaped polygon (loch or forest boundary), being buffered by 1km or less, then all squares **intersecting** with the buffer were assigned the given rating. For 4-figure grid references (1km square) being buffered by 1km or less, squares centred **within** the buffer were assigned the rating, as the central square containing the nest location still would be assigned the correct rating. For all buffers larger than 1km, squares for which the centre fell within the buffer were assigned the appropriate rating (for details of how the rule was applied for each species see Appendix 3, Table 1).

The 1km square scale for mapping the bird sensitivity ratings was considered a sufficiently fine resolution to be of use to local planning authorities and other decision makers. It is also the scale used by several other Geographic Information System (GIS) models to produce strategic locational guidance for renewable energy, e.g. Scotland's Renewable Resource (Snodin, 2001), the Highland Council Renewable Energy Strategy (Aquatera, 2006). Bird distributional data are available at different spatial scales and 1km square is the finest resolution available across a range of species, that also safeguards the majority of particularly sensitive bird records (see results).

Individual species' sensitivity maps were then amalgamated by choosing the highest sensitivity rating for each 1km square to produce the composite sensitivity map, with each of the 84 589 1km squares in Scotland allocated a sensitivity rating of 1 - high, 2 - medium or 3 - low or unknown, as above. Technical details of how the map was created are contained in Appendix 3.

The creation of the Sensitivity Map is envisaged as an iterative process with future updates as new data become available for the currently included, or additional, species, or as new research becomes available which prompts the revision of sensitivity criteria (see discussion).

Species' sensitivity criteria

Following reviews of literature on the foraging ranges, collision risk, disturbance distances and other relevant aspects of behavioural and population ecology, the following sensitivity criteria were developed (for full criteria see Appendix 1, for summary see Table 2).

Red-throated Diver *Gavia stellata*
Breeding sites used during and since the last national survey in 1994 were buffered by 1km, and these areas classified as 'high sensitivity'. This is based on best estimates of disturbance distances, and the high density of flight activity around breeding lochs (D. Jackson, pers. comm.). It was not possible to map flight corridors from breeding lochs to the sea or feeding lochs; these should be identified during the site-specific EIA process and incorporated in a risk assessment.

Black-throated Diver *Gavia arctica*
Lochs used for breeding within the last 10 years and lochs that are regularly used for feeding were buffered by 1km, and these areas classified as 'high sensitivity'. This is based on published disturbance-distances for Black-throated Divers (Lehtonen, 1970, Petterssen, 1985, Gotmark *et al.*, 1989), and on the fact that Black-throated Divers may feed away from their breeding lochs (Jackson, 2003). It was not possible to map flight corridors from breeding lochs to the sea or feeding lochs; these should be identified during the site-specific EIA process and incorporated in a risk assessment.

Slavonian Grebe *Podiceps auritus*
Lochs used for breeding between 1995 and 2005 were buffered by 1km and these areas classified as 'high sensitivity'. Breeding lochs used between 1990 and 1995 were buffered by 1km and these areas classified as 'medium sensitivity'. This is based on the high turnover of loch use by Slavonian Grebes (Summers *et al.*, 1994) and the fact that lochs are frequently recolonised after a period of absence (RSPB, unpublished data). The 1km buffers are based on best estimates of disturbance distances (see sensitivity criteria for other loch-breeding species). Areas of high flight activity outwith this area should be identified during the site-specific EIA process and incorporated in a risk assessment.

Bean Goose *Anser fabalis*
All fields used by Bean Geese in the Slammanan Plateau were mapped and buffered by 600m, and this area classified as 'high sensitivity'. This was based on the maximum reliably recorded disturbance distances from wind farms for geese (Kruckenberg & Jaene, 1999). This precautionary approach was considered appropriate due to the small, localised nature of the Bean Goose population; this site being the only one used by Bean Geese in Scotland (Simpson & Maciver, 2005).

Common Scoter *Melanitta nigra*
Common Scoter breeding sites used during the 1995 national survey were mapped. These were buffered by 1km, based on best estimates of disturbance distances, and this area classified as 'high sensitivity'. Common Scoter have been demonstrated to be particularly sensitive to disturbance, notably offshore, including in relation to wind farms (e.g. Mitschke *et al.*, 2001, Kaiser *et al.*, 2006). Data relating to disturbance onshore is lacking. Areas of high flight activity outwith the 1km buffer should be identified during the site-specific EIA process and incorporated in a risk assessment.

Red Kite *Milvus milvus*
All territory centres were buffered by 3km, roosts buffered by 5km, and these areas classified as 'high sensitivity'. This is based on the limited information available on Red Kite foraging ranges (Davis &

Davis, 1981, Ward, 1996, Orr-Ewing, pers. comm.), and on discussions with relevant species' experts. As with White-tailed Eagle, the small size and localised nature of the breeding population has meant that a particularly precautionary approach has been taken for this species.

White-tailed Eagle *Haliaeetus albicilla*
All territory centres, and any nest locations for which grid references were available, used since the re-introduction programme began, were buffered by 5km and this area classified as 'high sensitivity'. This is based on the limited information on the foraging range of the White-tailed Eagle (Masterov, Runde Conference 1996, Struwe-Juhl, 1996a, 1996b, Masterov, 2003), and was towards the upper end of the distances given, due to the small and localised the breeding population.

Hen Harrier *Circus cyaneus*
Sightings or nest locations from the 2004 national survey (the data do not allow differentiation between the two) were buffered by 2km and this area classified as 'high sensitivity'. This is based on literature on foraging ranges of Hen Harrier, which usually estimate a range of 1km to 2km radius (Schipper, 1973, Watson, 1977, Picozzi, 1978, Thompson-Hansen, 1984, Martin, 1987, Amar *et al*., 2004, Arroyo *et al*., 2005), and was at the higher end of this range as some of the data were just sightings and also as nest locations may move by up to 1km per year (Picozzi, 1984, Etheridge *et al*., 1997).

Nest locations/sightings from the 1998 national survey were buffered by 2km and this area classified as 'medium sensitivity'. This is so that areas are included for which there are historical records for the square but either no subsequent information or recent absence from suitable/previously used locations, requiring verification of status by site-specific data collection.

Golden Eagle *Aquila chrysaetos*
Nest locations from the 2003 national survey were buffered by a circle of radius 2.5km and this area classified as 'high sensitivity', and the area between this and a 6km buffer classified as 'medium sensitivity'. This is based on the "McGrady" model (used by Forestry Commission Scotland to determine the suitability of land for new woodland), which predicts that the core area of the range (the inner buffer) is where a Golden Eagle will forage for about 50% of the time, with the outer buffer representing the rest of the main range (McGrady *et al*., 1997, 2002). Thus, the buffering represents proportional use of the main home range.

In addition, nest locations from the 1992 national survey were buffered by 6km and this area classified as 'medium sensitivity'. This is so that areas are included where there are historical records for the square but either no subsequent information or recent absence from suitable/previously used locations, requiring verification of status by site-specific data collection.

Peregrine Falcon *Falco peregrinus*
Nest locations from the 2002 national survey were buffered by a circle of radius 2km and this area classified as 'medium sensitivity'. This is based on the fact that 70% of foraging behaviour is expected to occur within 2km of the nest (Weir, 1977, 1978).

Black Grouse *Tetrao tetrix*
Lek sites that were active during the 2005 national survey were mapped. Coverage of the survey was relatively low in terms of the Black Grouse's Scottish range, so additional data were collated from regional surveys in order to obtain as full coverage as possible. Lek sites were buffered by 1.5km and this area classified as 'medium sensitivity'. This is the area recommended for management for Black Grouse in the Species Action Plan (Anon., 2003), and the areas where Forestry Commission grants for Black Grouse management are targeted, and is based on literature on the size and distribution of

Black Grouse home ranges (Johnstone, 1969, Robel, 1969, Picozzi, 1986, Cayford & Hope-Jones, 1989, Cayford, 1990, 1993, Warren & Baines, 2004).

Capercaillie *Tetrao urogallus*
Areas used to identify stratified survey sites for the last Capercaillie national survey were plotted (Eaton *et al.*, in press). This was based on a compilation of records from conservation professionals, amateur ornithologists and landowners and managers, and is believed to accurately reflect current distribution (Eaton *et al.*, in press). Just the 'upper strata' were used: that is, sites where Capercaillie were present or breeding in 2003, and/or present in 2000. These were converted into forest blocks over 0.5ha, using the National Inventory of Woodland and Trees (NIWT (Forestry Commission, 2003)). These areas were classified as 'high sensitivity'.

The map of core areas used by the Capercaillie Biodiversity Action Plan Steering Group was not included on the sensitivity map, as it does not identify forest blocks, so would lead to a lot of unforested habitat being unnecessarily classified as 'high sensitivity'. This is not such a problem for the Capercaillie BAP Steering Group because they are dealing with a single species within a limited range and with very specific habitat requirements within their broadly identified area, whereas the sensitivity map is dealing with multiple species on a larger geographical scale.

Corncrake *Crex crex*
Calling males from the 1993 to 2003 annual surveys were mapped, given that the population is currently expanding, and that males use different calling locations in different years. These locations were buffered by 250m, as the nest is likely to be within this distance of the male's calling position (Green & Stowe, 1993). These circular buffers were then buffered by 600m and this area classified as 'high sensitivity', this distance being at the upper end of distances ranged by corncrakes during the breeding season (Hudson, Stowe & Aspinall, 1990), and also based on the literature on the maximum reliably recorded disturbance distance for birds in relation to wind farms (no information was available on disturbance distances for corncrakes).

Golden Plover *Pluvialis apricaria* and **Dunlin** *Calidris alpina*
For each of these species, mean tetrad counts from the New Atlas of Breeding Birds 1988-91 (Gibbons *et al.*, 1993) per 10km square were calculated. 10km squares with non-zero counts were then divided into five quantiles, and the 20% of 10km squares with the highest mean counts were classified as 'medium sensitivity'. In the case of Dunlin, taking the top 20% meant arbitrarily excluding equivalent counts in the next quantile, so the threshold was set at 22%. This approach was taken as coverage for these species did not allow for identification of important areas at a finer scale, and it was not thought justifiable to include data of this age as 'high sensitivity', thus the Atlas 'hotspots' were used to indicate areas where site-specific assessment may identify important areas for these breeding waders. It is expected that some additional areas of high sensitivity for these species in the uplands will be covered by the mapped high sensitivity areas for other species due to overlaps in range. Furthermore, inclusion of SPAs on the sensitivity map covers some high density areas of these species' distributions.

Arctic Skua *Stercorarius parasiticus*
1km squares holding ten or more birds during the Seabird 2000 (the last complete census of breeding seabirds in Britain and Ireland) were plotted and these were classified as 'high sensitivity'. This precautionary cut-off point was chosen, given that there were 2136 Apparently Occupied Territories (AOTs) during Seabird 2000 (Mitchell *et al.*, 2004), but there are thought to have been large declines even since then (Mavor *et al.*, 2005), and thus a cut-off at 21 AOTs (1% of the population found in Seabird 2000) was not thought sufficient.

Nightjar *Caprimulgus europaeus*
Locations of churring males from the 2004 national survey were buffered by 1km and this area classified as 'high sensitivity' and by 2.5km and the outer area classified as 'medium sensitivity'. This is based on the fact that Nightjars fly at turbine height when displaying or when pursuing prey (Cramp *et al.*, 1985, Palmer, pers. comm., Rolllie, pers. comm., Wotton, pers. comm.), and on literature on foraging flight distances for Nightjar (Glutz von Blotzheim, 1962, Schlegel, 1967, Bowden & Green, 1994, Alexander & Cresswell, 1996, Wichmann, 2004). Foraging distances can be greater than this, depending on the proximity of suitable foraging habitat to the nest, so site-specific assessment will be required to identify important foraging locations.

Chough *Pyrrhocorax pyrrhocorax*
Nest sites from the 2002 national survey (which are at the 1km square level of resolution) were buffered by 1km and this area classified as 'high sensitivity'. This is based on studies of foraging distances for breeding Chough (Holyoak, 1972, Bullock *et al.*, 1983, Bignal *et al.*, 1996, Cook *et al.*, 1999, Gray *et al.*, 2004, Whitehead *et al.*, 2006), and is in line with SNH guidelines for defining SPA boundaries, which use a distance of 1km from the nest site.

Special Protection Areas

These areas were classified as 'high sensitivity'; the application of sensitivity criteria for individual species above confirmed that this approach was justified for these species. SPAs were also used as surrogates for distribution and abundance data for coastal and breeding seabirds, wintering waterbirds and geese.

Results

The data layers for individual species are not presented here, due to the sensitive nature of the locations of many of the above species. The composite sensitivity map of Scotland comprises an amalgamation of single species data layers, in which the highest sensitivity rating for each 1km square was applied.

The map is presented here at the tetrad (2km x 2km) scale of resolution (Figure 1). The tetrad map is based on the 1km square map; if a tetrad contained any 'high sensitivity' 1km squares it has been shaded red, if it contained no 'high sensitivity' but one or more 'medium sensitivity' 1km squares it has been shaded yellow. Depth of colour depends upon the number of high/medium sensitivity squares within the tetrad. Unshaded areas of the map are of 'low/unknown' sensitivity. The tetrad map is intended for illustration. The underlying sensitivity ratings for 1km squares will be made available for strategic planning and to assist in determining individual proposals, under a data licence agreement, to protect a few locations of sensitive birds that are still apparent at the 1km square level.

Thirty seven percent of the area on the composite sensitivity map is classified as 'high sensitivity' (31 418 1km squares), thirty one percent as 'medium' (26 358 squares) and thirty two percent as 'low/unknown sensitivity' (26 813 squares).

Discussion

Map applications

The rapid increase in the number of wind farms proposed in Scotland has led to potential for conflict with bird conservation interests. It is important to minimise this conflict as many of the species of concern are protected under European Law, and are also an important part of Scotland's natural heritage, contributing significantly to its economy. A strategic approach could also help to minimise the wastage of resources that are currently going into proposing, assessing and objecting to inappropriately located wind farms.

The map is intended to provide a strategic view of the sensitivities of a suite of bird species in Scotland to onshore wind farm development and so facilitate locational guidance for wind farms to minimise conflict with bird species of conservation priority. The map is an indicative tool but does not replace the need for the standard site-specific checks: all categories of square should be subject to Environmental Impact Assessments, as appropriate.

The sensitivity map is based solely on bird sensitivities. However, it is recognised that local planning authorities and other decision makers will also need to take into account other factors, such as wind speed, grid connections, military low flying areas, cultural heritage needs and landowner and community interests. During the development of the sensitivity map, bird sensitivity ratings for the Highland region were made available for the Highland Renewable Energy Strategy. These ratings were fed into a GIS model, which took into account other factors e.g. wind speed, technical feasibility, cost, and a number of other planning constraints, such as designated sites and visibility (Aquatera, 2006). It is envisaged that other Local Planning Authorities may wish to make use of the map in similar ways.

Map limitations and possible solutions

The following caveats need to be taken into account when using the map:

- The map has been developed from the best information available. However, data deficiency means that the map cannot be comprehensive. Gaps in survey coverage for species on the map mean that there is no guarantee these species will not occur in 'low/unknown' sensitivity squares. Lack of available data also means that some sensitive species could not be included on the map. No liability is accepted for the presence or absence of species at particular sites contrary to that indicated on the map.

- The map is based primarily on the locations of breeding birds.

- The sensitivity map was created by collating data that were collected for other purposes, and thus not tailored specifically to the map's requirements (which would be an enormous task).

- The map is not a substitute for Environmental Impact Assessment, but is intended as an indicative map of likely bird sensitivities, to help guide decision-makers in the early stages of the planning process.

- The map will require updates to add new survey data for the species included, to add data for further species as they become available, and to reflect revisions to the sensitivity criteria in the light of new information, notably from research and experience at working wind farms.

The following section discusses further some of the limitations of the map, and ways in which these could be overcome in future updates of the map, as new sources of data become available.

Special Protection Areas

SPAs are designated to protect birds, but usually only cover core breeding and feeding territory. The birds may range more widely and therefore be affected by wind farms outwith the SPA boundary. It was not considered appropriate to buffer SPAs by a set distance, because different buffers would apply to different species and many SPAs have more than one of the species included here. However, SPA regulations or policies refer to the need to avoid adverse impact on the protected interest even when the development is outwith the designated area, and this should be considered during site-specific assessments. The sensitivity map will aid this process.

There are a number of SPAs for protected goose species which cover only the roosting areas, with important feeding areas for the birds on nearby farmland remaining undesignated. The Wildfowl and Wetlands Trust (WWT) is currently conducting a review of important feeding areas for protected goose species (R. Hearn, pers. comm.), however these data were not ready in time for inclusion in the sensitivity mapping project. It is envisaged these data could be an addition to later updates of the map. It should be emphasised that these areas are not included at present and should be identified during site-specific assessment to meet the underlying requirements of SPAs that there should be no adverse effect on the populations for which they were designated.

SPA data were used as surrogates for the distributions of several congregational species groups, notably non-breeding waterbirds and colonial breeding seabirds. Ideally, the map should be updated using distributional data which are more comprehensive for these species than the SPA network alone.

Updating the sensitivity map

The sensitivity map was created using the best data and information currently available. There will be a need to review and update the map as new data become available, as well as to review the species list and sensitivity criteria as new research comes to light. There are various surveys which will make useful updates or additions to the map, which were not available within the timescale of the current project, for example the 2006 national surveys for Black-throated and Red-throated Divers. There are also possible national surveys scheduled for 2007 for Merlin, Short-eared Owl and Common Scoter, and in 2008 for Osprey, which may be added. The last national survey of Merlin was in 1993-4. Plotting locations from this national survey alongside those from later regional surveys indicated that distribution had changed markedly, and so it was considered this dataset may not be suitable to represent current distribution. Not enough data are currently available to add Short-eared Owl to the sensitivity map, the most complete dataset being from the 1988-91 New Atlas of Breeding Birds (Gibbons *et al*., 1993), which was considered to be of limited value for this purpose, owing to the large spatial scale and the age of these data.

In the absence of more recent comprehensive data, Atlas data were used to identify areas which may contain high densities of breeding waders (see earlier). This approach was not considered suitable for mapping Merlin, Short-eared Owl and Osprey, as these species breed at much lower densities, and thus the 10km squares mapped would have included large areas of unoccupied habitat. For these species, the ideal data units would be locations of individual birds or nests. Fieldwork for the next breeding bird atlas is to be conducted in 2007-2011 (BTO, 2006), after which it will be possible to update the data for breeding waders, and so update the sensitivity map.

A field project to look at the effects of wind farms on a number of upland bird species is currently underway (L. Stephen, pers. comm.), and once results are available they will form a valuable source of information for updating sensitivity criteria, or possible amendments to the species list. Other similar research outputs will be useful for reviewing the sensitivity criteria.

Sensitivity criteria

The information available on which to base sensitivity criteria varied for different species. Even where information was good, time constraints meant that a best estimation had to be used to indicate the broad probability of use of an area. For example, for Golden Eagle, two different models exist to predict use of the area surrounding the nest. The first of these, the Research Information Note (RIN) model, was based on radio-tracking of Golden Eagles in Argyll, and found that 50% of foraging occurred within 2-3km of the nest, and 97% of foraging within 6km of the nest (McGrady *et al*., 1997, 2002). This model was refined to produce the Predicting *Aquila* Territories (PAT) model, which predicts percentage use of the range at a 50m x 50m pixel level, by modelling factors such as distance from the nest (at a finer scale than the RIN model), topography (for example, by taking into account preference for ridge features), elevation, habitat and regional eagle density (McLeod *et al*., 2002). This provides a more accurate picture of the range than the concentric ring buffer approach. However, to apply this model to all of Scotland's Golden Eagle population would have been beyond the scope of this project. Likewise, for other species where home or foraging ranges were estimated as ring buffers surrounding the territory centre, ideally these areas could be refined just to include habitat suitable for foraging, using satellite imagery. Again, this was outwith the scope of the project, and for some species the land use classes would not have been sufficiently refined to enable this approach to be used.

Future extensions

Roll-out to other countries

Scotland was considered to be a priority for a sensitivity map within the UK, as it has the highest number of onshore schemes currently being considered for planning approval. Many of these schemes are being proposed in the uplands, and Scotland's upland habitat supports important populations of birds of conservation concern. However, roll-out of the map to provide strategic locational guidance for other countries would be valuable. The Welsh Assembly is the only UK government to have issued strategic locational guidance for renewables, having identified seven broadscale 'Strategic Search Areas' (SSAs) in 2005 (Welsh Assembly, 2005). These SSAs are intended to identify large areas that are considered broadly suitable for wind farm development. Local Planning Authorities within Wales are then responsible for refining SSAs occurring within their area. The first Local Planning Authorities to do this were Denbighshire and Conwy, who produced a joint study that incorporated data supplied by the RSPB on bird sensitivities (Arup White Consultants, 2005). It would be valuable to build on these supplied sensitivity ratings, and to extend their supply to the other Local Planning Authorities. No current strategic locational guidance currently exists in England or Northern Ireland, and again extension of the map to these countries would be useful in contributing to such guidance.

SNH published strategic locational guidance for onshore wind farms in Scotland in 2001 (SNH, 2005), based on natural heritage considerations. These included landscape, designated sites, recreational interests and earth science interests, as well as sensitive bird areas. The map of sensitive bird areas was at the 10km square scale, and was based upon data from the New Atlas of Breeding Birds 1988-91 (Gibbons *et al*., 1993) for eight species of Annex 1 birds. A square was considered sensitive if four or more of the eight species were recorded as present within the 10km square. The project described here takes forward the mapping of sensitive bird species outwith designated sites by mapping sensitivities at a finer resolution, with more recent distributional/ abundance data, and incorporating buffers based on sensitivity criteria.

Offshore sensitivity map

Most of the currently installed capacity from wind energy is from onshore wind farms. However, with developments in technology, offshore wind farms are likely to make up a significant part of future wind farm development in Europe. For example, the UK government announced a major initiative to stimulate offshore wind farm development in July 2003 (Crown Estate, 2003). As offshore wind energy is still in its infancy, there is little information on the impacts of offshore wind farms on birds. Studies are underway, particularly in the Netherlands and Denmark, indicating a variable and site- and species-specific reaction as for onshore. Proposals for large wind farms in shallow sea areas may conflict with the feeding distributions of seabirds, notably seaducks, if there is a cumulative displacement effect. The lack of statutory marine protected areas makes identification of areas where offshore wind developments are likely to conflict with bird interests crucial. The identification of such areas and the development of a sensitivity map for the offshore areas around the UK are obvious potential extensions of the work presented here.

Wind farm capacity in Scotland

The map indicates that there is a greater incidence of bird sensitivities in northwest Scotland. Particularly sensitive areas occur in the Highlands, Western Isles and Northern Isles. Thirty seven percent of the area on the map is classified as 'high sensitivity' (31 418 1km squares), thirty one percent as 'medium' (26 358 squares) and thirty two percent as 'low/unknown sensitivity' (26 813 squares).

This indicative bird sensitivity map of Scotland is based only on bird sensitivities, without considering other potential constraints on wind farm development (for example, wind resource, cost, transmission, landscape). Capacity used in models to develop renewable energy strategies, currently tend to be based on four 2MW turbines per 1km square. This equates to a maximum potential capacity of 215GW in 'low/unknown' sensitivity squares alone, which may or may not be found to contain bird sensitivities or other factors that may constrain wind farm development. In reality, the maximum capacity is improbable. For example, removal from the calculations of 1km squares containing urban areas (taken from Land Cover Map 2000, Fuller *et al.* 2002) leaves 134GW capacity. However, given that a study for the Scottish Executive in 2001 (Snodin, 2001) estimated that 11.5GW of wind energy were available from less than 2% of Scotland's land area, at 3p/kWh generation cost, it seems likely that Scotland's renewable target of 6GW by 2020 (Scottish Executive, 2006), could be easily achieved without compromising the protection of sensitive bird species.

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Table 1. Data sources for species included on the sensitivity map

Species	Data Sources	Reference	Coverage
Red-throated Diver	1994 national survey	Gibbons *et al.*, 1997	Comprehensive on Shetland and Orkney, 11% of mainland breeding range
	All possible/probable or confirmed breeding records since 01/01/94 on Merlin (the RSPB internal library), supplementary records for Argyll, SNH data from monitoring of designated sites on the Uists, recent data submitted to the Rare Breeding Birds Panel.	RSPB data unit Roger Broad, pers. comm. Andrew Stevenson, pers comm. Malcolm Ogilvie, pers. comm.	Unknown
Black-throated Diver	Annual Black-throated Diver monitoring 1994-2004	For 1994 national survey see Whyte *et al.*, 1995	Comprehensive
Slavonian Grebe	Annual Slavonian Grebe monitoring 1990-2005	RSPB data unit	Comprehensive
Bean Goose	All fields used on the Slamannan Plateau	For methods see Simpson & Maciver, 2005	Comprehensive
Common Scoter	1995 national survey	Underhill *et al.*, 1998	Comprehensive
Red Kite	Annual Red Kite monitoring since beginning of re-introduction programme	RSPB data unit	Comprehensive
White-tailed Eagle	Annual White-tailed Eagle monitoring since beginning of re-introduction programme	Evans & Wilson, in prep.	Comprehensive
Hen Harrier	1998 national survey	Sim *et al.*, 2001	Comprehensive on Orkney, 77% of rest of Scottish population
	2004 national survey	Sim *et al.*, in prep.	Comprehensive on Orkney, 72% of rest of Scottish population
Golden Eagle	1992 national survey	Green, 1996	Comprehensive
	2003 national survey	Eaton *et al.*, in prep.	Comprehensive

Species	**Data Sources**	**Reference**	**Coverage**
Peregrine Falcon	2002 national survey	Banks *et al.*, 2003	92% of UK population
Black Grouse	2005 national survey	Sim *et al.*, in prep.	8% of range in Scotland/ northern England
	Black Grouse annual monitoring of core area by Perthshire study group	See Robinson *et al.*, 1993 for methods	Unknown
	Regional datasets for Perthshire, north of Scotland, Lothian and Borders, south west Scotland.	RSPB data unit	Unknown
Capercaillie	'Upper survey strata' used for the 2003-2004 national survey (Eaton *et al.*, in prep.)	RSPB data unit	Comprehensive
Corncrake	1993 national survey	Green, 1995	Comprehensive
	2003 national survey	O'Brien *et al.*, in prep	Comprehensive
	Annual Corncrake monitoring 1993-2003	RSPB data unit	Over 90% of Scottish population
Golden Plover	New Atlas of Breeding Birds 1988-91	Gibbons *et al.*, 1993	A minimum of eight tetrads in each 10km square of the range
Dunlin	New Atlas of Breeding Birds 1988-91	Gibbons *et al.*, 1993	A minimum of eight tetrads in each 10km square of the range
Arctic Skua	Seabird 2000	Mitchell *et al.*, 2004	Comprehensive
Nightjar	2004 national survey	Conway *et al.*, in prep.	54% of suitable habitat in Scotland
Chough	2002 national survey	Finney & Jardine, 2003	Comprehensive

Table 2. Summary of sensitivity criteria

Species	Data Used	Units	Spatial Resolution	Buffer zone	Sensitivity
Red-throated Diver	National Survey 1994, various regional datasets since 1994.	Possible/ probable/ confirmed breeding records	6 figure/ 1km squares	1km	1
Black-throated Diver	Surveys from last 10 years, knowledge of commonly used feeding lochs.	Apparently occupied territories	Loch boundary	1km	1
Slavonian Grebe	Breeding lochs used 1995-2005	Lochs with pair(s) present	Loch boundary	1km	1
	Breeding lochs used 1990-1995	Lochs with pair(s) present	Loch boundary	1km	2
Bean Goose	All fields used on Slammanan Plateau	Fields	Field boundary	0.6km	1
Common Scoter	National Survey 1995	Possible/ probable/ confirmed breeding records	6 figure	1km	1
Red Kite	All data since re-introduction	Nest locations	6 figure/ 1km squares	3km	1
	All data since re-introduction	Roost sites	6 figure/ 1km squares	5km	1
White-tailed Eagle	All data since re-introduction	All territory centres plus known nest locations	1km square	5km	1
Hen Harrier	National Survey 2004	Sightings/ nest locations	6 figure/ 1km squares	2km	1
	National Survey 1998	Sightings/nest locations	6 figure/ 1km squares	2km	2
Golden Eagle	National Survey 2003	Nest locations/ territories	6 figure/ 1km squares	2.5km	1
	National Survey 2003	Nest locations/ territories	6 figure/ 1km squares	2.5km to 6km	2
	National Survey 1992	Nest locations/ territories	6 figure/ 1km squares	6km	2

Species	Data Used	Units	Spatial Resolution	Buffer zone	Sensitivity
Peregrine Falcon	National Survey 2004	Nest locations/ territories	6 figure/ 1km squares	2km	2
Black Grouse	Various regional datasets, National Survey 2005	Lek sites	6 figure/ 1km squares	1.5km	2
Capercaillie	Upper survey strata used for the 2003-04 National Survey. Confirmed breeding/present in 2003, present in 2000	Forest blocks over 0.5ha containing Capercaillie since 2000	Boundaries of forest blocks over 0.5ha	0km	1
Corncrake	National Surveys 1993 to 2003	Singing males	6 figure	0.85km	1
Golden Plover	New Atlas of Breeding Birds 1988-91, mean tetrad counts	Adults	10km squares with top 20% of mean counts	0km	2
Dunlin	New Atlas of Breeding Birds 1988-91, mean tetrad counts	Adults	10km squares with top 20% of mean counts	0km	2
Arctic Skua	Seabird 2000	Colonies of 10 or more birds	1km square	0km	1
Nightjar	National Survey 2004	Churring males	1km square	1km	1
	National Survey 2004	Churring males	1km square	1km to 2.5km	2
Chough	National Survey 2002	Nest locations	1km square	1km	1
SPAs	All SPAs	SPAs	SPA boundary	0km	1

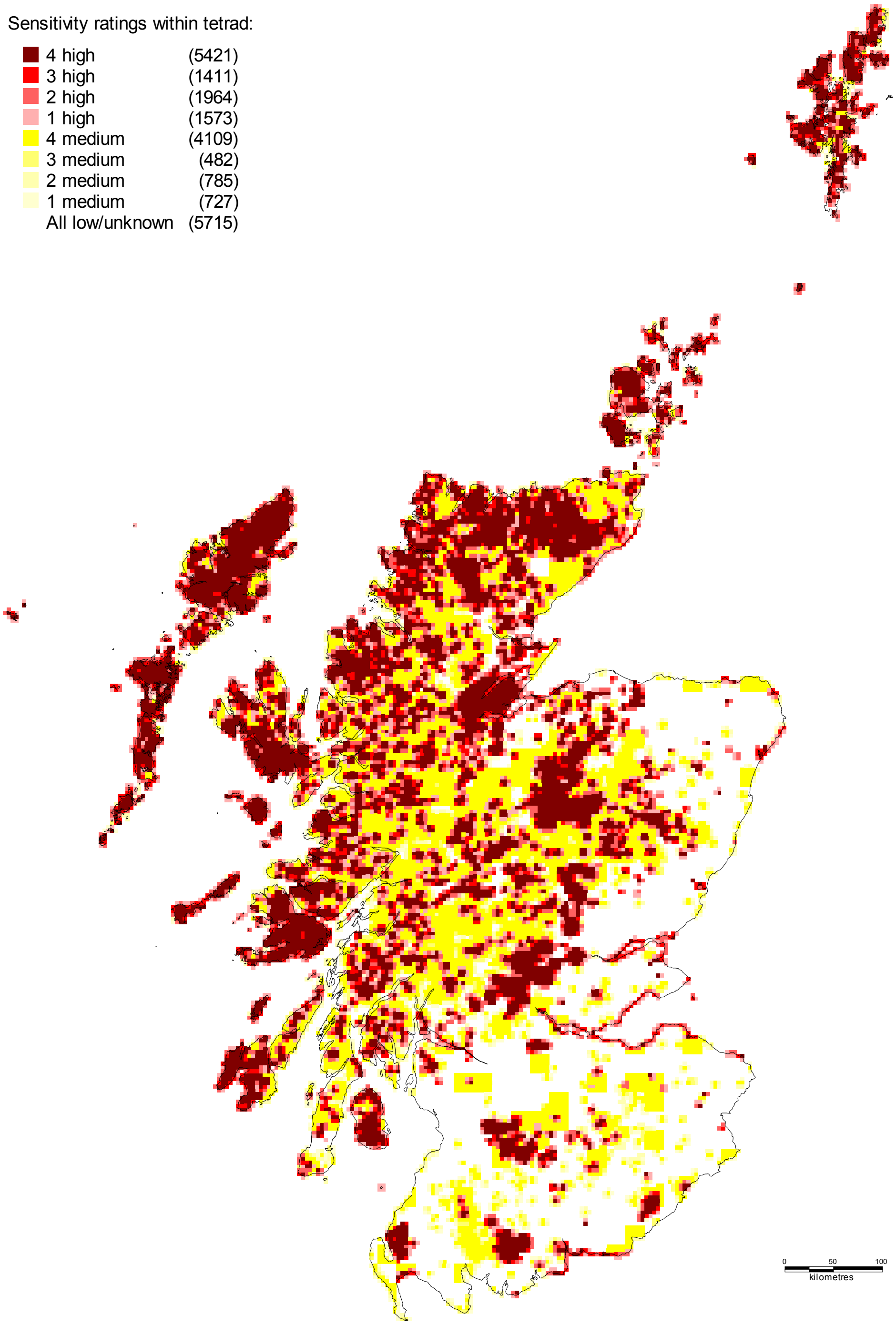

Figure 1. Composite Sensitivity Map of Scotland for location of onshore wind farms with respect to a suite of sensitive bird species (presented here at tetrad level of resolution to protect locations of sensitive species).

Appendix 1. Sensitivity criteria

1 Red-throated Diver

Introduction

The Red-throated Diver has a Holarctic breeding distribution extending up to 83°N and including the Boreal and Arctic zones of North America, Asia and Europe, with Britain being at the southern edge of the species' range (Gibbons *et al*., 1997). The British breeding and wintering populations are considered of international importance (Batten *et al*., 1990), with the British breeding population representing 1-2.5% of the European breeding population, which is in decline (Eriksson *et al*., 1988, Tucker & Heath, 1994). The British breeding population of Red-throated Diver is confined to north and west Scotland, 46% of this occurring in Shetland (Gibbons *et al*., 1997). Other important areas for Red-throated Divers are Orkney, which held 11% of the breeding population in 1994 (Gibbons *et al*., 1997), the Outer Hebrides, and Caithness and Sutherland (Sharrock, 1976, Gibbons *et al*., 1993).

The Scottish breeding population of Red-throated Divers is thought to have declined during the 19th century due to human persecution. The cessation of persecution and protection of breeding sites caused the population, although not the range, of the Red-throated Diver to increase in the early part of the 20th century (Saxby, 1874, Alexander & Lack, 1944, Venables & Venables, 1955, Parslow, 1967). The first national survey in 1994 estimated the British breeding population at 935 breeding pairs (Gibbons *et al*., 1997). General breeding distribution was similar in the 1968-72 and 1988-91 breeding bird atlases (Sharrock, 1976, Gibbons *et al*., 1993), although there was an eastward extension in the eastern Highlands and western Grampian (Dennis, 1984, Gibbons *et al*., 1993). The Red-throated Diver population in Shetland declined by over a third between 1983 and 1994, that in Orkney remained stable, and trends elsewhere are unknown (Gibbons *et al*., 1997).

In Shetland, the Scottish stronghold of the Red-throated Diver, the breeding population of Red-throated Divers has increased within the last hundred years (Venables & Venables, 1955), with there being a possible increase of 12-21% from the mid 1970s to the early 1980s, when Shetland held approximately 70% of the British breeding population (Gomersall *et al*., 1984). The reasons for the subsequent recent decline are not known. The decline has not coincided with a reduction in productivity, and is unlikely to be linked to predation or disturbance (Tharme *et al*., 1996). Increased mortality, reduced recruitment or an increase in non-breeding are all potential causes, and Tharme *et al*. (1996) suggest that an increase in the proportion of adults not breeding was the most likely explanation. Declines may have been linked to food shortage during the late 1980s (Tharme *et al*., 1996).

The Red-throated Diver is listed on Annex 1 of the EU Birds Directive, and is a species of medium conservation concern in the UK (Gregory *et al*., 2002). Forty-two percent of the British breeding population of Red-throated Divers occurs within the ten Special Protection Areas for this species (Stroud *et al*., 2001).

Breeding system and site fidelity

Breeding system

Red-throated Divers breed on small freshwater bodies. Birds usually arrive in early March and depart in September (Bundy, 1976). Following arrival, long inactive periods are spent on 'communal' lochs, where up to 15 birds have been counted (Bundy, 1976). First eggs are laid from mid May, with the peak laying period occurring mid May–mid June (Bundy, 1976, Tharme *et al*., 1996). Red-throated Divers are single brooded, but early losses are often replaced (Bundy, 1976, Tharme *et al*., 1996).

Incubation lasts 24-29 days (mean= 27) and young fledge after 38-48 days (mean= 43) (Cramp & Simmons, 1977). Nest sites found by Bundy (1976) were an average of roughly 50cm from the nearest water (n= 48).

Site fidelity and natal philopatry

Adults are site faithful and recruits demonstrate natal philopatry (Okill, 1992, Dickson, 1993). Ringing recoveries of birds in Shetland showed that males bred very close to their natal sites, on average within 2.2km (range 0.5- 4.4km, n= 5), whilst females dispersed widely, on average 37.8km (range 6-68km, n= 6) (Okill, 1992).

Age of first breeding in Red-throated Divers is not known but is thought to be 2-3 years (Cramp & Simmons, 1977). Ring recoveries from birds ringed in Shetland and Orkney found that first-winter birds were recovered as far south as central France (Okill, 1994). In their first summer, recoveries came from as far as Devon, Ireland and Norway, with a quarter of birds back in their natal area. In their second winter, birds were again widely distributed but by their second summer, two thirds of recoveries were in the natal area and the others were in northern Scotland. For older birds, winter recoveries remained widely dispersed, but all summer recoveries of adult birds came from within the areas of ringing. Nearly all birds left their native sites during the winter and were widely dispersed around the coast (Okill, 1994).

Once a bird had bred at a loch it rarely changed sites and if it did it would be to the nearest adjacent suitable loch (Okill, 1992). Twenty-four adults were retrapped on their breeding lochs on one or more occasion, with the longest period between initial capture and final retrapping being 13 years for a female and 12 for a male (Okill, 1992). Nearly all breeding adults were retrapped on their original breeding lochs, and the greatest distance moved was 1km. On Foula, Furness (1983) retrapped fourteen birds on their breeding lochs on one or more occasions, the longest period between first and last retrap being ten years. All birds were retrapped on their original breeding loch.

Disturbance and collision risk

Disturbance

Offshore at least, divers are particularly sensitive to disturbance, for example from approaching ships, and tend to occur mainly in marine areas with light sea traffic (Mitschke *et al.*, 2001, cited in Exo *et al.*, 2002). Garthe and Hüppop (2004) developed an index of sensitivity to marine wind farms for seabirds based on a number of factors (flight manoeuvrability, flight altitude, percentage of time flying, nocturnal flight activity, sensitivity towards disturbance by ship and helicopter traffic, flexibility in habitat use, biogeographical population size, adult survival rate and European threat and conservation status). Black-throated and Red-throated Divers were found to be the most sensitive of the species looked at.

Red-throated Divers also seem to be sensitive to disturbance onshore; breeding numbers of Red-throated Divers at a three-turbine wind farm site in Orkney declined as a result of disturbance during construction (Meek *et al.*, 1993). This result must be treated tentatively, however, as the sample sizes were small, with the decrease being from five pairs to two pairs. A study in Shetland found that breeding success was lower on waters with a high degree of disturbance, either by fishermen, peat diggers or bird watchers (Bundy, 1976). Norberg & Norberg (1971) considered that human disturbance (from anglers etc) contributed to five incidents of breeding failures. However, several studies have failed to find a correlation between breeding success and the distance from the nest to the nearest road (Shetland: Gomersall, 1986; Sweden: Norberg & Norberg, 1971, Dahlén & Eriksson, 2002). A disturbance-free distance of 300-900m from the nest is recommended for forestry workers (Currie & Elliot, 1997).

Foraging flights

Unlike other divers, which forage for their young in or near their freshwater nesting territory, Red-throated Divers usually nest on smaller lochs and fly to the coast to forage, or alternatively to the nearest large freshwater body if their breeding loch is further inland (Palmer, 1962, Davis, 1972, Bundy, 1976). Many foraging flights are made each day, including flights at dawn or dusk in poor visibility (Bergman & Derksen, 1977, Merrie, 1978, Andres, 1993, Eberl & Picman, 1993). Reimchen & Douglas (1984) found that adults made an average of 11 flights per day to the ocean, and that flights occurred throughout the daylight period, but extended into twilight during the first two weeks after chicks hatched. Merrie (1978) found that in Scotland, divers nested within 8km of the marine waters where they fed. Mean flight speed during foraging flights is 60.4km per hour (Norberg & Norberg, 1971). Migratory flight speed in *Gavia* species has been measured as an average of 77.9km per hour (Rydén & Källander, 1964). Observations of Red-throated Divers on Shetland found a high level of flight activity within 1km of breeding lochs (D. Jackson, pers. comm.). A large amount of this was by non-breeding birds.

There is little information on foraging flight height. Jim Williams (pers. comm.) states that on Hoy on a clear day Red-throated Diver are well above turbine height after about 1km, and only come closer to ground level as they approach the coast. This could change depending on weather and topography, however.

Collision risk

One of 48 cases where causes of death were specified for ring recoveries was killed flying into overhead wires (Okill, 1994). A review of the European literature found one reported casualty of a Red-throated Diver in Germany as a result of colliding with a wind turbine (Hötker *et al*., 2004).

Mapping Red-throated Divers

Data from the 1994 national survey were plotted. Survey coverage was comprehensive in Shetland and Orkney, with a sample survey of about 10% of breeding sites in the rest of Scotland being undertaken. Other data were collated in attempt to fill the gaps in coverage. Any possible, probable or confirmed breeding records of Red-throated Divers since the start of 1994 were downloaded from Merlin (the RSPB data library). Recent records submitted to the Rare Breeding Birds Panel were added. Supplementary records for the Uists were added from SNH surveys of designated sites, plus additional regional records for Argyll. Data were in the form of nest locations at either 1km square or 100m square resolution.

It was thought reasonable to include records since 1994, given the uncertainty as to recent trends in population and range for the Red-throated Diver, and the high site fidelity shown by Red-throated Divers to breeding lochs (Furness, 1983, Okill, 1992, Dickson, 1993).

Red-throated Divers are thought to be potentially susceptible to risk from both collision and disturbance (Langston & Pullan, 2003), although disturbance is considered to be the main concern. It is not clear at what distance disturbance effects will occur. However, in the study at Burgar Hill wind farm on Orkney where a decline in breeding Red-throated Divers occurred, and was thought to be a result of disturbance during construction, this was in contrast to an increase in numbers at a control site 2km away from the wind farm (Meek *et al*., 1993).

Given that Red-throated Divers travel up to 8km to forage, it was not thought practical to indicate foraging ranges on the map using a buffer, as the buffer would cover large areas not used by the divers (e.g. not towards the coast). Thus, Red-throated Diver nest locations were buffered by 1km and this area classed as “high sensitivity”. This distance was chosen as being between the 600m maximum reliable distance at which disturbance effects from wind farms have been demonstrated (albeit for geese, Kruckenberg & Jaene, 1999) and the 2km distance at which no effects of disturbance were apparent at Burgar Hill wind farm. It is also at the upper end of the disturbance-free distances suggested by Currie and Elliot (1997). Site-based assessment for any wind farms proposed near the buffer will need to take into account flight lines used by foraging birds as, although some feeding lochs have been included, and the buffers will cover some of the flight corridors, there are likely to be regular foraging flight lines not included here.

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These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in Scotland with respect to a suite of sensitive bird species. Thanks to Rhys Bullman, Ian Dillon, Pete Ellis, Richard Evans, Sam Gardner, Digger Jackson, Rowena Langston, Eric Meek, James Pearce-Higgins, Martin Scott and Jim Williams who provided helpful comments on an earlier draft of these sensitivity criteria.

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2 Black-throated Diver

Introduction

The Black-throated Diver is a widespread breeder across much of northern Europe, which accounts for less than a quarter of its breeding range. The European breeding population is relatively small (less than 92 000 pairs), and underwent a large decline between 1970 and 1990 (BirdLife International, 2004). Although most European populations were stable or increasing between 1990 and 2000, the species continued to decrease in Norway and its stronghold in Russia, and underwent a large decline of over 30% overall (BirdLife International, 2004). Thus, it is provisionally evaluated as 'Vulnerable' in Europe by the IUCN (BirdLife International, 2004).

The Black-throated Diver *Gavia arctica* is one of Britain's rarest breeding birds, and in Scotland breeds on large lochs almost exclusively in the north west of Scotland and the Outer Hebrides (Gibbons *et al*., 1993). The second national survey estimated the Scottish population at about 180 summering pairs (Whyte *et al*., 1995). This is an increase from the previous 1985 national survey estimate of 151 summering pairs (Campbell & Talbot, 1987). Whether this represents a population increase or was just a result of the higher survey intensity is unclear.

The 1985 national survey was the first comprehensive survey of the Scottish population, and records of Black-throated Divers prior to this are fewer and less reliable, so past population trends are difficult to deduce reliably. Mudge *et al.* (1991) collated historical records, and suggest that the Scottish population of the Black-throated Diver decreased substantially in the early part of the 20th century due to human persecution. A relaxation in persecution may have allowed an increase in numbers from the 1940s to the 1960s. There is no useful information on changes in the core mainland areas, but a southwestwardly extension (or possibly recolonisation) of the breeding range was noted between 1951 and 1974 when breeding pairs appeared in Arran, Ayrshire, Galloway and Central Scotland. They suggest that such an extension would be unlikely to occur if the overall population was in decline at that time. The evidence presented by Mudge *et al.* (1991) then points to another phase of decrease in the 1970s and 1980s, with losses being largely at the edge of the breeding range, and perhaps a consequence of the consistently low breeding success observed in these years (Mudge & Talbot, 1993).

There was no great change in the number of pairs attempting to breed in one of the core breeding areas in north west Sutherland in the 10-15 years leading up to 1984/5 (Talbot, Campbell & Dennis, 1984, Talbot & Campbell, 1985).

Differences in coverage make comparisons between the 1985 national survey (Campbell & Talbot, 1987) and the 1968-72 breeding bird atlas (Sharrock, 1976) difficult. Campbell & Talbot (1987) suggest that there was no great change in distribution between the two, especially considering the sporadic nature of breeding in some of the marginal areas in the southern part of the British range and the difference in survey methods. However, Mudge *et al.* (1991), based on records throughout the 1980s, suggested a reduction of up to 27% in the number of 10 km squares occupied. Jackson (2005) suggests that although Mudge *et al*. (1991) document a slight range contraction in south west Scotland, the historical information is too poor to show whether the Scottish Black-throated Diver has ever been substantially greater than it is at present.

Jackson (2005) modelled loch use by Black-throated Divers and found that it is likely there are sufficient suitable lochs to accommodate a moderate increase in the Scottish population. Monitoring data support this, as in the past 15 years a number of lochs with no previous history of breeding have been colonised and the national population appears to have increased slightly (Whyte *et al*., 1995). This follows a project of provisioning nesting rafts that has boosted productivity (Hancock, 2000).

The Black-throated Diver is listed under Annex 1 of the EU Birds Directive, and is a species of high conservation concern in the UK (Gregory *et al*., 2002). A high proportion (61%) of the Scottish Black-throated Diver population breed on Special Protection Areas (Stroud *et al*., 2001). It is a long-lived species reproducing at a low annual rate (Nilsson, 1977).

Territories and ranging behaviour

Black-throated Divers hold exclusive territories that can comprise part of a loch, a whole loch, or two or more nearby lochs (Mudge & Talbot, 1993). Detailed studies in west Sutherland found that Black-throated Divers show a high degree of site-fidelity from year to year (Mudge *et al*., 1991). Over half of the sites known to have been occupied before 1900 were still occupied in 1985-9 (Mudge *et al*., 1991). At least 23 of 30 pairs studied in north west Sutherland used nearby alternative sites from time to time (Talbot, Campbell & Dennis, 1984). Most breeding lochs are occupied by a single pair, although some large lochs have 2 or 3 pairs. One very large loch (Loch Maree) is occupied by up to 10 pairs (Jackson, 1997).

Young are fed on fish and/or invertebrates. Jackson (2003) found that the majority of provisioning is done following attendance foraging (i.e. with chicks), with some excursion foraging, usually fairly soon after hatching. The nature of this excursion foraging changed as chicks got older. On days 1-8 foraging adults normally hunted on the nesting lake, stayed mostly within 500m of the brood (which nearly always remained at the nest site) and generally swam back with the prey. Later on, in the post-brooding phase, excursion foraging was commonly further afield, either to more distant parts of the breeding lake, or to another lake or area of sea (typically within 2km) and the adult usually flew back with any prey (Jackson, 2003).

Disturbance

Offshore at least, divers are particularly sensitive to disturbance, for example from approaching ships, and tend to occur mainly in marine areas with light sea traffic (Mitschke *et al*., 2001, cited in Exo *et al*., 2002). Garthe and Hüppop (2004) developed an index of sensitivity to marine wind farms for seabirds based on a number of factors (flight manoeuvrability, flight altitude, percentage of time flying, nocturnal flight activity, sensitivity towards disturbance by ship and helicopter traffic, flexibility in habitat use, biogeographical population size, adult survival rate and European threat and conservation status). Black-throated and Red-throated Divers were found to be the most sensitive of the species looked at.

The Scottish Black-throated Diver population is of conservation concern due its small size and low breeding success. The chief causes of breeding failure are due to flooding or predation of nests at the egg stage (Mudge & Talbot, 1993). It is often difficult to ascertain the primary cause of nest failure, as predation may follow disturbance. Disturbance, particularly from anglers but also recreation more generally, is a potential threat to the Black-throated Diver (Thom, 1986, Mudge, Talbot & Wilts, 1987). Human disturbance may enable predation (Petterssen, 1978, Mudge, Talbot & Witts, 1987, Gotmark *et al*., 1989) or lead to desertion (Mudge, Talbot & Witts, 1987).

Gotmark *et al*. (1989) found that flushing distances of incubating Black-throated Divers in Sweden and time spent off the nests were related to stage of incubation and the recreational use of lakes. They measured flushing distances in response to their approaching boat, and found that it varied between 0 and 750m. Mean flushing distance was roughly 300m +/- 200m in the early stage of incubation, and 200m +/- 300m in the later stages. Median distance was 310m in the early stage and 80m in the late stages of incubation, and based on this, Gotmark *et al*. (1989) suggested that the current disturbance-free zone around Black-throated Diver sanctuaries in Sweden of 100m was too small. Even greater flushing distances have been recorded in Sweden in the 1960s (Lehtonen, 1970). Median flushing distances of 100m for early incubation and 400m for late incubation were found for Lake Sottern in Sweden in 1983 (Petterssen, 1985). It is possible that these distances could be lower than those found by Gotmark *et al*. due to habituation. Currie & Elliot (1997) recommended a 300-900m disturbance-free zone around breeding sites, depending on the time of year, for forestry work, although it is not clear what this is based on.

Collision Risk

Black-throated Divers fly strongly but with poor manoeuvrability (Jackson, 1997), meaning that as well as the risk of disturbance from wind farms, they may also be susceptible to collision risk.

Mapping Black-throated Divers

Given the fact that the Scottish population may be increasing at present, and recolonising previously used or new sites (Jackson, 2005), lochs that have been used for breeding (or form part of a breeding territory used) between 1994 and 2004 have been mapped. Lochs which are considered to be important feeding lochs have also been included.

These lochs have been buffered by 1km, and this area rated as "high sensitivity". This buffer is based on the fact that the maximum flushing distance from a boat was 750m (Gotmark *et al*., 1989), and the recommended disturbance-free distance for forestry is 300-900m, and the considered view that a wind farm could potentially cause more disturbance to breeding divers than either of these. Site-based assessment for any wind farms proposed near the buffer will need to take into account flight lines used by foraging birds as although some feeding lochs have been included, and the buffers will cover some of the flight corridors, there are likely to be regular foraging flight lines not included here.

Acknowledgments

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in Scotland with respect to a suite of sensitive bird species. Thanks to Rhys Bullman, Dave Butterfield, Richard Evans, Sam Gardner, Mark Hancock, Digger Jackson, Rowena Langston, James Pearce-Higgins and who provided helpful comments on an earlier draft of these sensitivity criteria.

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3 Slavonian Grebe

Introduction

The Slavonian grebe *Podiceps auritus* is one of Britain's rarest breeding birds (Batten *et al*., 1990). It colonised Scotland in 1909 (McGhie, 1994) and since then has bred in the Highland, Grampian and Tayside regions, usually nesting in moderately sized lochs in beds of bottle sedge (Thom, 1986). These lochs are vacated in the winter months. Annual censuses have taken place since 1971 and the population initially increased to reach 60-80 pairs from 1978 to 1993 (Perkins *et al*., 2005). The population then declined during 1994-2000, falling to just 31 pairs in 2000, the lowest level since counts began (Perkins *et al*., 2005). The number of occupied lakes also halved over the same period to 15 in 2000 (Perkins *et al*., 2005). The use of a few regular breeding lochs (for example around half of the population currently nests at Loch Ruthven), combined with the restricted distribution makes the population vulnerable to adverse management changes at these lochs. Due to the population decline and its low numbers, the Slavonian grebe is a species of medium conservation concern in the UK (Gregory *et al*., 2002), and is protected under Annex 1 of the EU Birds Directive.

There has been a high turnover in the breeding lochs used by Slavonian Grebes (Summers *et al*., 1994). Sixteen lochs were occupied in 2005 (RSPB, unpublished data), eleven (69%) of these being occupied by a single pair, two lochs by two pairs and the remaining three by four, six and nineteen pairs respectively. Thus, one loch held 43% of the breeding population, and three lochs combined held 66% of the breeding population. There are six Special Protection Areas for Slavonian grebes, and these hold 53% of the UK breeding population (Stroud *et al*., 2001).

Breeding productivity of Scotland's Slavonian Grebe population is low compared to that in Scandinavia and Iceland (Fjeldså, 1973, Ulfvens, 1989, Crooke *et al*., 1993). This could be a result of a number of factors such as rapid fluctuations in water levels, human disturbance, changes to the food supply associated with introduction of predatory fish such as pike and changes in water quality associated with catchment land use (Crooke *et al*., 1993). Perkins *et al.* (2005) investigated the impact of nest predation rates on productivity, but found that predation rates were not particularly high. Wave damage was a major cause of clutch failure at Loch Ruthven, the main breeding loch. However, recent work has indicated that the Slavonian Grebe's population changes, erratic site occupancy and variable productivity in Scotland may all be linked to global climate and that the factors listed above may be of less importance than previously thought (Benn, pers. comm.).

Disturbance, Collision Risk etc

Breeding System

Slavonian Grebes usually start to arrive on breeding lochs in early April (Summers & Mavor, 1995). The earliest eggs are laid in early May, with laying occurring until early August (Summers *et al*., 1994). Birds roost on their breeding lochs. Birds start arriving at moulting sites from the end of June onwards (Summers *et al*., 1994), but the median time for adults to depart from breeding lochs is not until late July/early August (Summers & Mavor, 1995). Loch Ruthven and two other lochs formed the main moulting sites in 1993, with some birds probably moulting at their breeding lochs (Summers *et al*, 1994). The number of birds at moulting lochs peaks in August and then stays level for the next three weeks or so before declining as grebes begin to move to the coast, but with some birds not leaving the moulting lochs until the end of October (Summers *et al*., 1994).

Disturbance

Lochs occupied by Slavonian Grebes in 1992 and 1993 had more visitors than lochs which were not occupied, suggesting that disturbance did not affect loch choice (Summers *et al*., 1994). The median distances to roads and houses from lochs occupied by Slavonian grebes were 313m (range 2-225m), and 475m (range 200-1538m) respectively. These distances were not significantly different (and in fact were smaller) than those for lochs not occupied by Slavonian grebes (Summers *et al*., 1994). Distances at which Slavonian grebes flushed from nests when approached both by boat and on foot were also measured. Grebes flushed at an average distance of 10m when approached on foot and took 2 minutes to return. This distance was lower (8m) when disturbance levels were high, and was larger (up to 30m) at more remote lochs, with birds taking longer to return. When approached by a boat, birds did not flush until the boat was an average of 6.4m away (n= 7). There was no effect of use by either boat or bank fishers on loch selection by Slavonian grebes, however productivity was negatively correlated with use of lochs by bank fisherman (Summers *et al*., 1994). Thus, although disturbance does not seem to have an impact on loch selection by Slavonian grebes, it may have a negative effect on productivity.

A buffer zone of 50-150m around Slavonian grebe nests is recommended for forestry workers in Great Britain (Currie & Elliot, 1997).

Collision Risk

There is little published data on movements or flight heights of Slavonian Grebes. However, colour-ringing of Slavonian Grebes in Scotland since 2001 shows that there is quite a lot of movement between sites at all stages of the season (Benn, pers. comm.). This movement probably takes place at night. Thus, although Slavonian Grebes were not considered at particular risk of collision in Langston and Pullan's review (2003), this new evidence would suggest they may be at risk.

Mapping Slavonian Grebes

Lochs used between 1990 and 2005 were mapped. This is because of the high turnover of loch use by Slavonian grebes and as lochs are frequently recolonised after a period of absence. Slavonian grebes may be at risk of both disturbance and collision during the breeding season. Although disturbance does not seem to affect choice of breeding site by Slavonian grebes, it may have a negative effect on productivity. There is no information with regard to the distance from particular sources of disturbance (such as a wind farm) at which this effect may occur. Likewise, there is little information published on movements of Slavonian Grebes during the breeding season. In line with other loch breeding species, and following consultation with species experts, a buffering distance of 1km was decided upon. Due to the fact that loch use is erratic, and appears to be becoming increasingly so (Benn, pers. comm.), lochs used for breeding between 1995 and 2005 will be classified as 'high sensitivity' and those used between 1990 and 1995 as 'medium sensitivity'.

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4 Bean Goose

Introduction

The Bean Goose is widespread and abundant across the Palaearctic, breeding from Scandinavia to eastern Siberia (Madge & Burn, 1988). There are five races of Bean Goose. Two of these occur in the western Palaearctic; the Taiga Bean Goose (or Western Bean Goose) *Anser fabalis fabalis*, and the Tundra Bean Goose *Anser fabalis rossicus*. Birds wintering in Britain are nearly all Taiga Bean Geese, except for a recent population of about 50 Tundra Bean Geese at North Warren.

Both races breed at high latitudes eastwards from Fenno-Scandia, Taiga birds generally breeding further south and west than Tundra Bean Geese (Hearn, 2004). Most of the population of Taiga Bean Geese winters in the Baltic region (southern Sweden and Denmark, with smaller numbers in other neighbouring countries), and these birds, which originate from Finnmark and Russia, have increased in numbers in recent times (Parslow-Otsu, 1990). Smaller, generally decreasing numbers, from Scandinavian breeding grounds, winter on the North Sea plain (with approximate numbers being Jutland: 200-800 at five sites, Netherlands: 800-2 000 at eight sites, Britain: 400 at two sites)(Parslow-Otsu, 1990). The Bean Goose is not an internationally threatened species, and with a total world population of around 90 000 to 110 000 individuals (Hearn, 2004) the British population of around 480 is relatively tiny (Simpson & Maciver, 2005). However, there have been dramatic declines in the breeding ranges and numbers of Bean Geese in Norway and Sweden throughout most of this century due to persecution and habitat changes, and in this context the British wintering population assumes greater importance (Parslow-Otsu, 1990). Parslow-Otsu (1990) estimated the total Bean Goose population in central Scandinavia to be only of the order of 3000 individuals, with 17% of these wintering in Britain.

In the first half of the 19th century, the Bean Goose was a common winter visitor to northern Britain and East Anglia, being one of the most numerous of goose species wintering on the British Isles (Sutherland & Allport, 1994). A widespread decline began from the 1860s onwards, with most populations severely diminishing or disappearing by the 1920s (Parslow-Otsu, 1990). Since 1940, there has been further decline in numbers, and there are now only approximately 480 Bean Geese (427 Taiga Bean Geese) wintering in Britain. These are restricted to three regular sites; the two oldest being in the Yare Valley in Norfolk (Dunmore, 2005) and the Slamannan Plateau in Central Scotland (Simpson & Maciver, 2005), with a third recent populations of 53 birds occurring at North Warren (RSPB, 2005). A peak population of 112 birds were also counted at a site near Lowestoft in Norfolk, however, these birds are likely to hail from the Yare Valley population (Dunmore, 2005, Strudwick, pers. comm.). The Slamannan Plateau site has been occupied since 1985/6 (Hearn, 2004). Birds wintering at the two older British sites are likely to hail from different areas of Sweden, with colour rings observed on some of the Scottish population showing that at least some of the geese are from a breeding population established in south west Sweden as a result of a reintroduction programme (C. Mitchell *in litt.*, cited in Parslow-Otsu, 1991). In addition, the Slamannan population arrives on the wintering grounds 6– 8 weeks earlier than the Yare population (late September in comparison with mid November). All other colour-ringed birds found in Britain indicate links with the Scandinavian breeding population (Parslow-Otsu, 1991). Until recent years, the Yare Valley was the most important of the British sites (Owen, Atkinson-Willes & Salmon, 1986). Recently, the Scottish population has been increasing, from around 130-140 birds in 1993/4 (Smith, Bainbridge & O'Brien, 1994) to a peak of 300 birds in 2005/6 (Trubridge, pers. comm., with the peak for 2004/5 being 262 birds) and now constitutes over half of the total British population (Simpson & Maciver, 2005). The Yare Valley population, meanwhile, is currently decreasing (Hearn, 2004).

The Bean Goose is a species of medium conservation concern in the UK (Gregory *et al.*, 2002) and is listed under Annex II/I of the EU Birds Directive. The Yare Valley Bean Goose site is a Special Protection Area (Stroud *et al.*, 2001) and the Slammanan Plateau population is currently a proposed SPA.

Collision Risk, Disturbance etc

Hearn (2004) described lack of site protection and unfavourable changes in land use as the principle threat to the Slamannan Plateau population of Bean Geese. Fraser (2003) identified visitor pressure, recreational activity and primary industry operations in the vicinity of the main feeding areas resulting in disturbance, and other developments that may increase the potential for disturbance, as two of the main threats to the Slamannan population of Bean Geese.

There are several reliable studies of geese indicating negative effects up to 600m from wind turbines, i.e. reduction in bird use of, or absence from, the area close to the turbines (e.g. Pink-footed Geese, European White-fronted Geese, reviewed in Langston & Pullan, 2003). Geese are prone to both disturbance and collision (Pink-footed Geese, European White-fronted Geese, Barnacle Geese, Brent Geese, reviewed in Langston & Pullan, 2003).

Moorehead and Epstein (1985) identified large wetland birds, such as geese, as being especially susceptible to collisions with wind farms. A review of the European literature found one reported casualty of a Bean Goose in Germany as a result of colliding with a wind turbine (Hötker *et al.*, 2004).

Average avoidance distances of 100m for wind turbines in lines and 200m for turbines in clusters have been observed in Pink-footed Geese in Denmark (Larsen & Madsen, 2000). Avoidance behaviour by European White-fronted Geese in Germany was more marked, with studies showing substantially lower densities of feeding geese found within 600m of the wind turbines (Kruckenberg & Jaene, 1999). Bean Geese on the Slamannan Plateau have been found to select areas more distant from buildings and roads (Smith *et al.*, 1994).

Pink-footed and Greylag geese wintering in north east Scotland tended to avoid the proximity of roads when foraging on agricultural land. Flocks were found a median distance of 400m from the nearest road (100-1100m), and not within 100m (Keller, 1996). Avoidance of roads by geese has also been shown for Pink-footed Geese in Denmark by Madsen (1985) and for Bean Geese and White-fronted Geese in Germany by Mooij (1982). Mooij found that feeding intensity was lower 250m from quiet roads and 400m from heavily used roads. In central feeding areas, intensity was severely reduced within 250m of a road or other source of disturbance, irregular between 250-350m, ‘frequent’ from 350-450m, ‘regular’ from 450-550m but only ‘intense’ over 550m from sources of disturbance (Mooij, 1982). Madsen found that distances from roads below which grazing intensity was reduced corresponded with flight distances shown by flocks when approached by a car. Keller (1996) only recorded a few flight distances, as most flocks were found at distances where they did not react to cars, however, flight distances seemed to be shorter than those found by Madsen. Keller (1996) found that flight distances ranged from 100-250m (mean= 190m, n= 6), whereas Madsen found that flight distances were about 500m in autumn and 3-400m in spring. It was suggested this could indicate habituation, as geese in the study area in Scotland were exposed to much higher traffic density than those in Denmark. In Scotland, at distances where geese did not show any reaction to passing cars they often reacted to cars stopping on the road or other kinds of disturbance such as tractors or walkers.

The presence of low power lines has been shown to reduce foraging intensity of White-fronted Geese and Bean Geese in the adjacent 40-80m in western Germany, although no effect was observed for lines over 60m high (Ballasus & Sassinka, 1997).

Bean Geese, and particularly western Bean Geese, are much more wary than other geese, and inhabit places with low levels of human disturbance throughout their wintering range (Parslow-Otsu, 1990). Increased levels of disturbance (from traffic and hikers) have been suggested as a reason for the abandonment of a historically occupied site in Ayrshire in the 1930s (Parslow-Otsu, 1990).

Mapping Bean Geese

Fields used regularly by the Slamannan Plateau population of Bean Geese were mapped. Given the small population of Bean Geese which winter in Scotland, the fact that disturbance distances for Bean Geese are poorly quantified, and the fact that Bean geese are thought to be particularly wary of disturbance, a precautionary 600m buffer was applied to the fields used by Bean Geese. This represents the higher end of disturbance distances reported for geese (European White-fronted Geese) in response to wind turbines. 1km squares within this buffer was classified as 'high sensitivity'.

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5 Common Scoter

Introduction

The Common Scoter *Melanitta nigra* has a Holarctic breeding distribution stretching from southern Norway to eastern Russia, favouring high latitude tundra, with small outlying populations in Iceland, Britain, and Ireland (Scott & Rose, 1996). The first Scottish breeding record for the Common Scoter was in Sutherland in 1855, with breeding recorded in adjacent areas in the late 19th century and more widely in Scotland in the next century (Baxter & Rintoul, 1953, Thom, 1986). The Scottish population breeds on small moorland lochans in the north of Scotland and larger lakes with wooded islands in central Scotland (Steele & Mellon, 1998). The Rare Breeding Birds Panel collates data on the Common Scoter, however these are mainly casual records. The first comprehensive survey occurred in 1995, which estimated 95 breeding pairs in Britain, being largely restricted to northern and western Scotland (Underhill *et al*., 1998). These pairs were located at sixty-one sites in twenty-three 10km squares, with nine of these 10km squares holding 83% of the British breeding population (Underhill *et al*., 1998). The Flow Country in Caithness and Sutherland held most breeding sites, but the south and west Highlands were equally important in terms of numbers of breeding birds present. Over half of the occupied sites (52%) in Britain held only one or two individuals, but notable concentrations were recorded at two sites in the south and west Highlands (42 and 23 birds respectively) and on one site on Islay (24 birds).

The lack of comprehensive surveys prior to the 1995 national survey makes population trends in Britain difficult to determine. The 1995 figure was the largest estimate to date, but the fact that birds were absent from previously occupied areas; Shetland, Central Scotland and Dumfries and Galloway, suggests that the population may be in decline (Underhill *et al*., 1998). Numbers on the Flow Country declined from 50 to 35 pairs between 1988 and 1991 (Hancock, 1991), and more recently there is evidence of decline in numbers and population in the Flow Country following regular monitoring on Forsinard reserve and adjacent areas, of areas that were checked in the 1980s (Hancock, pers. comm.). Underhill *et al.* (1998) suggest that the Scottish population has been declining in numbers and range over the past 30 years. Gregory *et al.* (2002) report a decline of 37% between 1973/77 (RBBP data) and 1995. Sites surveyed during the 1995 national survey were revisited in 1996 for the RSPB Species Action Plan, and this showed a population decline of 19% for Scotland.

The Common Scoter occurs mainly in marine habitats outside the breeding season, and it undergoes an extensive moult migration in Europe. To this end, large concentrations, often exceeding 10 000 birds, assemble in the western Baltic, the eastern North Sea, and offshore in Britain and Ireland and western France (Hagemeijer & Blair, 1997). Between 50 000 and 65 000 Common Scoter winter in UK waters, mostly on fairly small, traditional sites, with only 13 of these holding 1000 or more birds (Cranswick, in press). Lack of systematic surveys at these sites makes it hard to determine trends, but it is considered that numbers may have been relatively stable at most sites in recent decades (Cranswick, in press). Other estimates for numbers of Common Scoter wintering around Britain and Ireland include 25 000-30 000 (Campbell, 1986), or at least 35 000 (Prater, 1981), although Cranswick (in press) suggests these are underestimates due to lack of data. Kirby *et al.* (1993) used these estimates to suggest that a maximum of 4% of the total western Palaearctic population winter in British and Irish waters. Icelandic-ringed birds have been recovered in Britain (Kirby *et al*., 1993), but as this population numbers only around 500 pairs (Cramp & Simmons, 1977) most of our wintering birds must come from breeding areas in Fennoscandia and western Siberia (Owen *et al*., 1986). Kirby *et al*. (1993) suggest that Common Scoters are ‘far less common than they once were’.

The European breeding population accounts for less than a quarter of the Common Scoter’s global breeding population. This European population is considered to have been stable between 1970 and

1990, and populations in most countries were stable during 1990 to 2000, so the overall population is considered stable and is given a 'Secure' status in *Birds in Europe* (BirdLife International, 2004a), and a favourable conservation status in *Birds in the European Union: a status assessment* (BirdLife International, 2004b). The global breeding population size is estimated as 100 000 to 130 000 pairs (BirdLife International, 2004a). Thus, only a tiny proportion of the breeding population occurs in Britain. However, the issue is one of preserving range and diversity of the Common Scoter rather than just numbers. There are a number of threats to Common Scoter in the UK, notably oil spills, with two major spills in the 1990s, one of which killed at least 4000 Common Scoter (Cranswick, in press). Afforestation, water quality, competition for food from introduced roach and predation by American mink are all threats to breeding birds (Underhill *et al.*, 1998, Cranswick, in press). Shell fishing, recreation, aggregate dredging and proposed offshore wind farms are all considered potential threats to non-breeding birds, which congregate in large, dense flocks, although the possible impact of these activities is largely unknown (Cranswick, in press).

The Common Scoter is on the 'Middle List of Globally Threatened/Declining Species' in the UK Biodiversity Steering Group Report and is listed on Annex II/2 and III/2 of the EU Birds Directive. The Common Scoter is a species of high conservation concern in the UK due to the rapid contraction in its breeding range (Gregory *et al.*, 2002), making it the only red-listed duck species. Common Scoter no longer breed in Northern Ireland (Gittings & Delany, 1996). The British breeding population is relatively well protected by SSSIs in the Flow Country and on Islay, but in the west and south Highlands 75% of occupied sites holding 40% of the British population are not protected (Underhill *et al.*, 1998). Forty nine percent of the British population occurs on two multi-species Special Protection Areas (Caithness and Sutherland Peatlands, Rinns of Islay), with the rest occurring outwith SPAs (Stroud *et al.*, 2001).

Breeding system

Common Scoters usually arrive on their breeding lochs from late April to early May (Loch Lomond: Mitchell, 1976; Flow Country: Hancock, 1991; Lower Lough Erne, Ireland: Steele & Mellon, 1998). During May, the Scoters move freely from island to island, but by mid-May pairs have formed and movement is more localised (Hancock, 1991, Steele & Mellon, 1998). After mating, males return to the sea, and most have left by the end of June (Hancock, 1991, Steele & Mellon, 1998). Egg laying occurs mainly late May to late June (occasionally later), and hatching occurs from early to late June (Hancock, 1991). Clutch sizes at Lower Lough Erne were between 4 and 9 (mean= 6.7, Ferguson, 1971). Common Scoters are single-brooded, with replacement clutches smaller than first ones. Incubation takes 30 to 31 days (Cramp & Simmons, 1977). Cramp and Simmons (1977) record that the male defends an area around the female but that little is known about home range and territories, and this appears still to be the case (Smith, in prep.).

Nests in Scotland are usually built amongst the heather on an island, or close to a small loch or pool system in hilly moorland (Ferguson, 1967). Common Scoter sometimes nest along river courses and use corridors as nursery areas for raising young (Fox & Bell, 1994). Some broods are raised in pool systems with chicks moving between small pools. Nests are usually near water, typically around 20m from the shore in Scottish moorland areas; nests further from water may be reached by overgrown drainage ditches (Sharrock, 1976).

Wintering

The Common Scoter is almost exclusively marine outside the breeding season, wintering (October to March) and moulting (mid-July to October) offshore in large flocks on shallow coastal waters, typically 500m-2km offshore (Cramp & Simmons, 1977). The Common Scoter is a diving duck that feeds on submerged benthic prey, principally bivalve molluscs, and so prefers water depths of less than 10m earlier in the winter, extending to up to 20m as the winter progresses. Water depth is the main factor that determines distance offshore (Kaiser *et al.* 2006). The main influx of Common Scoter to British waters occurs in October and November, with peak numbers usually from December until early February (Campbell, 1986). Lack (1986) shows a winter distribution around the entire British and Irish coast except steep and rocky sections, with higher numbers in northern and western Scotland. Very large flocks are concentrated on relatively few sites, mostly near the mouths of large estuaries and in shallow bays. The timing and direction of movements, other than arrival at and departure from breeding areas are largely unknown.

Disturbance and collision risk

Disturbance

Scoters are considered particularly sensitive to disturbance offshore, e.g. from approaching ships, and tend to occur in marine areas with light sea traffic (Mitschke *et al*., 2001, cited in Exo *et al*., 2002, Kaiser *et al*., 2006). Garthe and Hüppop (2004) developed an index of sensitivity to offshore wind farms for 26 species of seabirds, which was based in part on sensitivity to disturbance by ship and helicopter traffic. They rated Common Scoter as the most sensitive of the 26 species to these forms of disturbance. Offshore wind farms are often cited as a potential threat to non-breeding Common Scoters, due to this sensitivity to disturbance and the fact that they aggregate in dense flocks. Fox (2003) states that quasi-predator stimuli (such as human disturbance) can modify the local distribution of scoters. Shaw (1975) mentions three pairs being displaced from their favoured small bays on Inchlonaig by boats anchoring overnight.

Flushing distances

Modelling work was conducted by COWRIE (Collaborative Offshore Wind Research Into the Environment) in the UK to predict changes in over-winter mortality of Common Scoter offshore resulting from disturbance displacement due to existing, consented and proposed wind farms in Liverpool Bay (Kaiser *et al*., 2005). Flushing distances of Common Scoter in response to the approach of a 35m long ship were measured, and it was found that small flocks flushed at 1km and large flocks at 2km. Ten of the 59 flocks observed flushed at a distance greater than 2km. Kaiser *et al*. (2005) concluded that Common Scoter were extremely wary of shipping, as reported elsewhere in their European range.

Avoidance: Disturbance displacement

Research has been conducted in Denmark looking at effects of offshore wind farms on Common Scoter; this was used to help decide what parameter estimates to enter into the COWRIE model. Results from a study at Tunø Knob wind farm in Denmark were inconclusive, but indicated a possible disturbance effect for Common Scoters (Guillimette *et al*., 1998). They found that numbers of both Common Scoter and Common Eider decreased in the wind farm area compared to the reference area in the two years following construction. It was suggested this might have been due to fluctuating food supply and so not to do with the wind turbines, however only partial recovery of Common Scoters was noted indicating possible displacement by the wind turbines (Guillimette *et al*., 1999).

The National Environmental Research Institute (NERI) has conducted research on the effects of two other wind farms in Danish coastal waters; at Nysted and Horns Rev wind farms (Christensen *et al*., 2004, Kahlert *et al*., 2004, Petersen *et al*., 2004, Petersen, 2004). At Horns Rev, no Common Scoter were observed within 1.5km of the wind farm, and avoidance was shown within an area of 2km and 4km of the wind farm.

Avoidance: Flight paths

Radar tracking at Horns Rev showed that flocks of sea birds changed flight paths to avoid wind farms at a distance of 0.4km or 1km (depending on the direction of approach), and that few bird flocks entered the wind farm. Those that did entered almost parallel with the turbine rows. The distance from the wind farm at which number of flight tracks began to decline with increasing proximity to the wind farm was 1.5-2km and 2km (again depending on direction of approach). Thus, 2km may be the distance at which flying flocks of seabirds respond to wind farms. Flight track data from Nysted showed that birds avoided the wind farm both during the day and at night. Common Scoters wintering offshore fly at night, but Tulp *et al*. (1999) found that flight activity was reduced in the vicinity of a wind farm, with a diminishing effect observed 1000-1500m away from the nearest turbine. Birds approaching parallel to the alignment of the turbines were more likely to cross them than if the approach was perpendicular to turbine alignment, indicating a barrier effect can operate. Generally, the closer offshore wind turbines are to the shore the more likely they are to intercept flight paths by birds such as scoters moving to feeding areas.

COWRIE: Conclusions

Both at Horns Rev and Nysted there seems to be some indication that sea ducks tend to exhibit avoidance of wind farms. At both sites, Common Scoter were not seen within the wind farm and at Horns Rev, none were seen within 1.5km of the wind farm.

Given these results, the COWRIE review used as its worst-case scenario avoidance zones of 2km around wind farms in their models. Using this value, addition of one particular proposed wind farm to the model indicated that it would cause a significant adverse effect on the overall over-winter mortality of the population. Kaiser *et al*. (2005) stated that 'Common Scoter are extremely sensitive to any form of disturbance'.

Onshore research

Onshore, there has been less research into the effects of disturbance. However, Mitchell (1977) and Partridge (1987) implicate increasing disturbance from recreational activity in causing decreases at Loch Lomond and Lower Lough Erne respectively. Disturbance is also considered to have been important in driving duck declines in Iceland (Bengston, 1972, in Partridge, 1987). Cramp and Simmons (1977) noted that Common Scoter are generally 'rather wary'. A safe working distance for forestry workers of 300-800m from Common Scoter nest sites is recommended in Britain (Currie & Elliot, 1977).

Collision risk

There is little published information on flights of Common Scoter during the breeding season. Common Scoters prefer high pH lochs, and females and their broods have been observed moving to lochs of higher pH than the nesting lochs a few weeks after hatching (Fox *et al*., 1989). Many studies (Partridge, 1987, Partridge & Smith, 1988) have found evidence (such as the prompt movement of males during incubation, and of pre-fledging broods, to the sea) suggesting that food supply may be limited in fresh water compared to the sea. Males only spend about two months at breeding areas (Mitchell, 1977, Partridge, 1987, Partridge & Smith, 1988). Thus, during these flights it is possible that Scoters could be at risk of collision. There is a lot of movement before the breeding season in the Flow Country, mainly in April, when birds are seen away from breeding sites, and make regular flights between the main breeding lochs (Hancock, pers. comm.). Movements of up to 5km are probably common during this period (Hancock, pers. comm.).

The Bern report considered Common Scoter as potentially at risk of collision, barrier effects, disturbance displacement, and direct habitat loss from wind farms. However, the Danish studies of offshore wind farms suggest that Scoter are not very susceptible to collision; and that flying birds generally, and sea-ducks in particular, respond behaviourally to wind farms at a distance of between 1 and 3km by altering flight tracks such that only a few birds fly through the wind farms (discussed above in the 'Avoidance: flight paths' section), and if they do enter a wind farm do so parallel to the turbine alignment. Only 10 of 36 000 Common Scoter at Horns Rev wind farm were found within the wind farm area. Kaiser *et al*. (2005) states that 'in contrast to some other types of birds e.g. raptors and vultures, it is unlikely that collision of sea-ducks with wind turbines is a serious source of mortality. There are no substantive reports in the literature that would, as yet, indicate this is a problem for common scoter'. Flight altitudes for Common Scoter offshore were between 0 and 8m (S.D.= 5.2m, n= 2), and mean flight speed was 86.1km/hr, (range 46.9-86.1, S.D.= 10.7, n= 11).

There was little information on flights of breeding Common Scoter, or their response to onshore wind farms.

Mapping Common Scoter

The Common Scoter was chosen for inclusion in the sensitivity map despite it not being an Annex 1 species, due to its red-listed status, the fact that it has a very small, localised breeding distribution, and that it is very sensitive to disturbance. It is also a long-lived species, which does not start breeding until 2-3 years old, and with low annual productivity, thus losses will have more impact on the overall, small, population.

Literature reviewed here would suggest that the main likely impacts of wind farms on Common Scoter would be due to loss of habitat due to disturbance displacement and that collisions would be rare.

Based on the research looking at responses of Common Scoter to wind farms and other forms of disturbance *offshore* (in the absence of detailed information onshore), it would seem that Common Scoter respond to offshore wind farms behaviourally at a distance of 1 to 2km from the wind farm. However, caution must be taken before applying these distances to the behaviour of breeding Common Scoter to onshore wind farms, as birds are likely to be much more tied to a site during the breeding season. There is little information on, for example, disturbance distances for breeding birds, on which to base a buffer zone. Thus, it was decided to buffer Common Scoter locations from the 1995 national survey by 1km, in line with the buffers applied for other loch-breeding species, and for this area to be rated as "high sensitivity". Site-based assessment for any wind farms proposed near the buffer will need to take into account flight lines used by foraging birds as generalised buffers were not felt to be appropriate for this species.

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6 Red Kite

Introduction

The Red Kite was once an abundant resident breeder throughout most of Britain, but became extinct except in mid Wales by the late 19th century due to persecution (Carter *et al.*, 1998). The Welsh population reached a low of three pairs in 1905, but has slowly increased to at least 160 pairs today (O'Toole *et al.*, 2000). The global Red Kite population has also suffered as a result of persecution, loss of habitat and pollution and is now very fragmented and restricted to relatively small areas of the former range (Evans *et al.*, 1994). Ninety percent of all Red Kites breed in France, Germany and Spain (Gensbol, 1989, Evans & Pienowski, 1991, Meyburg *et al.*, 1991).

In 1989, a Red Kite re-introduction programme began. Between 1989 and 1993, 93 Red Kites were imported from Sweden and released near Inverness (O'Toole *et al.*, 2000), with successful breeding occurring in 1992, and a breeding population of 38 breeding pairs by 2005 (B. Etheridge pers. com.). A similar re-introduction programme was established in southern England, with 93 Red Kites imported from Spain being released between 1989 and 1994 (O'Toole *et al*., 2000) also leading to a successful breeding population with 109 pairs in 2000 (Wotton *et al*., 2002). In 1996, a second Scottish release site was established in central Scotland, and 103 Red Kites from Germany were released between 1996 and 2000 (Wotton *et al*., 2002, D. Cameron, pers. comm.). Two more release sites have also been established in England in the Chilterns and Yorkshire.

The rate of increase of the introduced populations has been extremely rapid (37% for England, 42% for Scotland, 1994-7), but the range is just expanding slowly from the core areas, with just the odd pair being found over 50km from the nearest core area (Carter *et al*., 1998). The first co-ordinated national survey in Britain was undertaken in 2000 and yielded 430 breeding pairs; seven of which were in central Scotland and 33 in north Scotland (Wotton *et al*., 2002). At the beginning of the re-introduction programme, the Red Kite was one of only three globally threatened species occurring in the UK (Collar & Andrew, 1988). Following increases in parts of central and northern Europe, however, this classification has been reduced to near threatened (BirdLife International, 2005). It is also a species of medium conservation concern in the UK (Gregory *et al*., 2002), and listed under Annex 1 of the EU Birds Directive. The Red Kite is a qualifying species for just one Special Protection Area, which is in Wales and holds 15 pairs of Red Kites, or 9% of the British breeding population (Stroud *et al*., 2001).

Home ranges, site fidelity etc

Territory size

The Red Kite has a single brood but replacement laying occurs occasionally (Carter *et al*., 1998). A mean area per breeding territory of 15km^2 has been estimated from RSPB data (equivalent to a circle with radius 2km; Ward, 1996). Red Kites in Wales may forage up to 10km or more away from the nest or roost (Davis & Davis, 1981), but often remain much closer.

Densities

Some pairs in England bred within a few hundred metres of their nearest-neighbours, and seven pairs bred within an area of about 4km^2 (Carter, 1997).

Breeding distribution of the north Scottish population has expanded from six to ten 10km squares with the number of breeding pairs within the core 10km square having increased from three in 1994 to nine pairs in 2000 (Wotton *et al*., 2002). Mean breeding density per 10km square in north Scotland has increased from 1.3 pairs per 10km square in 1994 to 3.3 in 2000 (Wotton *et al*., 2002). The central Scotland population began to breed in 1998, and although most attempts have been within a few kilometres of the release cages, the population is now distributed over eight 10km squares (Wotton *et al.*, 2002, D. Cameron, pers. comm.).

Dispersal

As with many large raptors, the Red Kite shows a high degree of natal philopatry and despite the tendency for a proportion of young birds to disperse, the majority return to breed in the area where they are fledged or released (Evans *et al.*, 1999, Newton *et al.*, 1994). The re-introduced populations in England and Scotland tend to form communal roosts within the core breeding areas, and these are almost exclusively used by first or second year birds, with the adults roosting within their territories (Carter *et al.*, 1998), although some territorial adults also make occasional visits to main roosts during winter (D. Cameron, pers. comm.).

Seventy nestlings were tagged in Wales between 1975 and 1979, and it was found that individuals moved up to 22km between birthplace and breeding place, but having bred, most stayed in the same locality from year to year (Newton *et al.*, 1989). There were 49 incidences of birds staying on the same nesting territory in subsequent years, and only four of birds changing territories, in which case the new territories were 5, 3, 2 and 26km from the old territories (Newton *et al.*, 1989). Nest sites of birds from the recolonisation programmes were not fixed and pairs shift to new nests within a territory or to new territories between years (Evans *et al.*, 1999). The distance moved by pairs between nesting attempts ranged from 0 to 55km, but was usually less than 6km (Evans *et al.*, 1999). In England, between 1992 and 1995, 56% of successful pairs returned the following year to breed again in the same nest (Evans *et al.*, 1999). In Scotland in the same period, Evans *et al.* (1999) reported that only one successful breeding pair out of 15 returned to the same nest it had used previously. However, it is now considered that Red Kites in the north of Scotland and extremely site faithful, with almost all successful breeding pairs re-using the same nest the following year (B. Etheridge, pers. comm.).

Disturbance, Collision risk etc

Disturbance

Nests can be built close to sources of disturbance; for example one nest was built about 150m from the nearest occupied building, however this nest failed at the egg stage (Carter, 1997), and some of the kites in Scotland have chosen to nest on farms and in large gardens close to people. A pair in 2005 nested only 40m from an occupied farmhouse and reared two young (B. Etheridge pers. comm.). However, Davis & Newton (1981) attributed a minimum of 9% of all recorded nest failures to disturbance near the nest through agricultural or other human activity. This was also the proportion of nest failures due to disturbance in Wales, when 11 nests probably failed due to incidental disturbance (Carter *et al.*, 1998). A small number of nests in England and Scotland are also thought to have failed as a result of disturbance (Carter *et al.*, 1998). A disturbance free zone of 400-600m around occupied Red Kites nests is recommended in Britain (Petty, 1998).

Collision Risk

Large birds of prey may be susceptible to risk of collision with power lines or wind turbines. There is currently very little information on the problem in the UK and it is not known whether power lines pose a serious threat to raptors at the population level. Carter *et al.* (1998) suggest that collision with power lines is likely to be at most a local problem, but at least four kites from the English re-introduction project and four from the Scottish project have so far been found dead under power lines. Evans *et al.* (1994) found that 13% of all recorded deaths were probably due to collision with electricity wires and high-speed vehicles (persecution accounting for 47% and just 13% of recorded mortality being from natural causes). Davis and Newton (1981) give similar figures for the period 1951-1980; of 32 Red Kites reported dead, 31% died due to human persecution, 9% due to collision with electric wires or vehicles, and 19% from natural causes.

A Black Kite *Milvus migrans*, was one of 51 raptor carcases found under a wind farm consisting of 256 turbines as Tarifa in Spain, which had a very high raptor mortality rate (SEO/BirdLife, 1995). A review of the European literature found forty recorded Red Kite fatalities as a result of collision with wind turbines in Germany since 1989, with two further casualties occurring in the UK and one in Spain (Hötker *et al*., 2004).

Mapping Red Kites

All Red Kite territories used since the re-introduction programme began will be mapped and buffered by 3km, and this area classified as 'high sensitivity' given the fact that the estimated breeding range of the Red Kite is roughly equivalent to the area of a circle with radius of 2km and Red Kites range more widely in winter (Orr-Ewing, pers. comm.). Roost sites will be mapped and buffered by 5km, given the large number of birds that use them. This precautionary approach is justified on the basis of the small, localised population of Red Kites in Scotland.

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7 White-tailed Eagle

Introduction

At the beginning of the 19th century, the White-tailed Eagle *Haliaeetus albicilla* was widely distributed as a breeding bird along the north and west coasts of Scotland and Ireland with an estimated 200 breeding pairs (Love, 1988), and the distribution in Britain was probably even wider in earlier times (Love, 1983). The White-tailed Eagle became extinct in Scotland in the early 20th century due to persecution (Love, 1983), with the last nesting attempt occurring on Skye in 1916 (Baxter & Rintoul, 1953). No natural recolonisation took place within Britain. This was because the species had also suffered in Europe from persecution, habitat loss, and latterly pesticides (Love, 1988).

A population has been re-established in western Scotland by the release of 82 wild-bred birds imported from Norway between 1975 and 1985 on the Isle of Rum (Green *et al.*, 1996). The reintroduced White-tailed Eagles first bred in the Hebrides in 1983 and reared the very first chick two years later (Love, 2003). Since then, the numbers of breeding pairs and young fledged have slowly increased (Evans *et al.*, 2003). Following the recommendations of population modelling based on the demographic data (Green *et al.*, 1996), a further 58 eagles were released from a mainland site further north in Wester Ross between 1993 and 1998 (Love, 2003) and now over 30 pairs are breeding in the highlands and islands of Scotland (Evans & Wilson, in prep.). The range has expanded north, south and west of the release points in Rum and Wester Ross, and is expanding at a rate of roughly 8% annually (Evans *et al.*, 2003). Although the Golden Eagle is more aggressive than the White-tailed Eagle, the White-tailed Eagle seems to be experiencing little difficulty in re-establishing, some Golden Eagles being ousted completely but others managing to live alongside their larger cousin (Love, 2003). There are also areas available to White-tailed Eagles where Golden Eagles are absent, thus prospects are good for continued growth and re-expansion of the population (Love, 2003).

During the last two decades, populations of the White-tailed Eagle have also been recovering or have recovered in many areas of its northern and central European range (Dennis, 2003). Range recovery in Norway, Sweden and Germany has been occurring at a rate of approximately 10km, 10km and 6km per annum over the last 20 years (Dennis, 2003). The European breeding population is still small, however, perhaps less than 5 000 pairs, and as such the species is evaluated by the IUCN as ‘Rare’ in Europe (BirdLife International, 2004).

The White-tailed Eagle is a species of high conservation concern in the UK (Gregory *et al.*, 2002) and is listed on Annex 1 of the EU Birds Directive. There are no Special Protection Areas designated for White-tailed Eagle in the UK (Stroud *et al.*, 2001).

Collision Risk, Disturbance and Buffer Zones

Collision Risk

White-tailed Eagles are known to collide with wind turbines, with Krone and Scharnwerber (2003) reporting collisions with wind turbines in north-eastern Germany of an adult female in January 2002 and a sub-adult male in April 2002. These are two of a total of thirteen fatalities recorded in Germany due to collision with wind turbines since 1989 (Hötker *et al.*, 2004). Given that there are 470 breeding pairs of White-tailed Eagles in Germany (in comparison with an estimated 30 breeding pairs in Scotland; Evans & Wilson, in prep.), Hötker *et al.* (2004) estimated that the total German population was probably around 1 400 White-tailed Eagles. Given that many corpses will not be found, Hötker *et al.* (2004) gave the example that ten wind farm related fatalities of White-tailed Eagles per year in Germany would increase the background mortality rate by 0.7%.

Construction of a wind farm on the island of Smøla in central Norway began in 2002. The island is an Important Bird Area with an estimated breeding population of 65-70 pairs of White-tailed Eagle (Folkestad, 2006). To date, nine White-tailed Eagle fatalities due to collision with wind turbines have been recorded since construction on phase 2 of the wind farm was completed in summer 2005; six adults and three juveniles (Follestad, pers. comm.). Most corpses were incidental finds; formal corpse searches only being implemented in 2006. Five of these collisions were detected within just one month during the breeding season of 2006.

In Sweden, of 58 White-tailed Eagle deaths reported between 1989 and 1994,
14 were due to collisions with cables and wires and electrocution and 9 were due to collisions with trains (Helander, 2003). Electrocution is also said to 'affect White-tailed Eagles greatly' in Northern Belarus (Ivanovsky, 2003). Anthropogenic factors such as collisions with trains and cars, collisions with power lines, electrocution and poisoning are presently the main causes of death in Germany, with collision with wires accounting for 7% of 120 deaths, and electrocution for 9% (Krone *et al*., 2003). In a study in Norway, cause of death was mainly trauma including electrocution through collision with power lines (~67%, n= circa 100, Runde Conference, 1996).
Of 14 White-tailed Eagle deaths in Greenland, six were caused by 'unspecific trauma' (Krone *et al*., 2004).

Disturbance distances

The White-tailed Eagle is generally considered a shy bird when breeding, avoiding human settlements and activities, and nesting in remote areas (Collet, 1921, Hagen, 1952, Glutz *et al*., 1971, Haftorn, 1971, Fisher, 1982). According to Gerdehag & Helander (1988) the distance between White-tailed Eagle eyries and inhabited houses and some other human activities in Sweden was at least 1km, and Koivusaari *et al*. (1988) state that human disturbance is one of the two most important problems for the White-tailed Eagle in the Kvarken area of Finland. Human disturbance is also considered one of the main limits to the population in the Ukraine at the moment (Gavrilyulk & Grishchenko, 2003). However, there was little work to quantify the effect of human activities on breeding White-tailed Eagles prior to work in Norway by Folkestad (2003).

Lots of work has been done looking at disturbance distances in Bald Eagles. Bald Eagles may avoid human disturbed areas when selecting nesting and foraging sites. A study of behavioural responses to human activity at the nest found a distance of 72m caused a response in Bald Eagles, and that eagles did not habituate, but instead flushed at increasing distances, with additional disturbances (Fraser *et al*., 1985). Frenzel *et al.* (1985) meanwhile found that there was no effect of human activity on nesting success of Bald Eagles in Minnesota within a distance of 500m. Other studies however, state that in both the Bald Eagle and the White-tailed Eagle, regular human activities are less disturbing than irregular activities (Grubb *et al*., 1992, Folkestad, 2003). Recommendations of no-activity zones within 500-600m of the nest are recommended based on studies of Bald Eagles in Michigan as beyond this distance response frequency was generally below 30% (Grubb *et al*., 1992).

The minimum distance from permanently established White-tailed Eagle eyries to public roads and permanently inhabited houses in the Møre og Romsdal County in Western Norway is roughly 500m (Folkestad, 2003). The birds may settle close to human activity, occasionally very close, but normally pairs will adjust the nest site in relation to human activity and only occasionally hold permanent nest sites nearby (Folkestad, 2003). Nest sites within a territory on Mull were 30m and 70m from a forest road (MacLennan & Evans, 2003). White-tailed Eagles in the Ukraine are tolerant to minor disturbance near fields, pastures and rarely used roads, and the distance from such places is usually 0.5-1.5km (mean= 0.56 +/- 0.3, n= 24). In contrast, nests are further from permanent sources of disturbance such as villages and regularly used roads (mean= 2.4km, +/-0.1km, n= 24). In the Alesund/Runde area of Norway, where 45 new territories have become established since 1982, the majority are within 1km of human habitation: the closest distance from occupied nest to occupied house is 120m (Folkestad, Runde conference, 1996). Analysis of 97 White-tailed Eagle nests in Finland and Sweden found that 50% of them were within 500m of the nearest road, and 30% within 500m of the nearest summer cottage. These relatively short distances from nests to roads etc are often cited as evidence of the White-tailed Eagle's tolerance to human activity. However, it is usually not mentioned how long these territories are occupied for or the breeding success of the pairs nesting in them.

Folkestad (2003) found that White-tailed Eagles nesting close to human activities normally have a remarkably low productivity compared to those with inaccessible or remote nests. This was most pronounced at Smøla in Norway, where productivity was reduced within a distance of 1km from public roads and inhabited houses, compared with the productivity of nests beyond this distance. A study of White-tailed Eagle nests in the Quark area in Finland also showed that disturbance can reduce productivity; 68% of nests built in new locations succeeded compared to just 15% of nests built in traditional locations (Koivusaari *et al.*, 1988a). This difference was attributed to people not being aware where the new nests were, and the resulting lack of disturbance by curious members of the public (Koivusaari *et al.*, 1988a, 1988b). Folkestad (2003) concluded from his study in Norway that the White-tailed Eagle is unable to adapt to close human disturbance in the breeding grounds and that these should be protected. In light of this, Folkestad stated that 'Along the Norwegian coast, the development of large scale wind farms may be the most serious threat to White-tailed Eagles in years to come'. The presence of wind farms themselves may cause disturbance to White-tailed Eagles, but they are also associated with an increase in numbers of people, both during construction and also during operation when the network of access roads for maintenance teams are also available to other users, thus potentially making formerly remote areas more accessible.

A study of the effects on White-tailed Eagles of the construction and operation of the wind farm on Smøla in Norway found that, despite the presence of 34 nest sites for 14 to 18 pairs of eagles prior to construction, no successful breeding had occurred within the perimeter of the wind farm (Folkestad, 2004). The nearest successful breeding in 2004 was 800m from the nearest planned turbine. Some breeding activity has been observed just 32m from a turbine for two years after construction, but this was not successful. Since this report, two successful breeding attempts occurred in 2005 and one pair has attempted to breed in 2006 (Folkestad & Follestad, pers. comms.). This is a substantial reduction in breeding pairs compared with the situation prior to construction of the wind farm. Breeding success across the rest of the island has also been the lowest since monitoring began 31 years ago, since the turbines were erected (Folkestad, 2004). Folkestad (in litt.) estimated that breeding success is reduced within a distance of 1km from turbines (Folkestad, in litt.), which is similar to that in relation to other built development (Folkestad, 2003).

Disturbance-free zones

Year-round buffer zones around White-tailed Eagle nests, which are protected from all human activity, are recommended in Estonia (Randla & Tammur, 1996). Likewise, year-round buffers of 200m, extending to 500m in the breeding season, are protected from forestry activity in forests in Poland (Mizera, 2003). Year-round buffers of 100m, extending to 300m in the breeding season, are protected in Germany (Hauff, 1996), and a 500m breeding season buffer exists in Iceland (Skarphedinsson, 2003). In Lithuania, the nest tree and a surrounding zone with radius 100m are protected (Mecionis & Dementavicius, 2003). In Latvia, some protection zones of 400-800m exist (Lipsbergs & Bergmanis, 2003). In Finland, an area of 50m around the nest is fully protected from forestry, with only minor operations being permitted within 500m of the nest, and none in this area during the breeding season, and building of permanent roads within 1km of a nest is not permitted (Stjernberg, 2003). On the Aland Islands of Finland, polygonal protection zones exist, with diameters ranging from 0.2 to 3.6km, mostly in the range 0.4 -1.4km (Stjernberg, 2003).

Ranging behaviour

Site fidelity

White-tailed Eagles are fairly site faithful, with studies of the birds released on Rum suggesting that pairs use the same two or three nest sites, alternating between them, with six of the ten pairs studied having alternative eyries (Love, 1988). Of the birds released on Rum, some immatures wandered widely, but settled on fixed home ranges within 100km of the release site, usually at 2-4 years old (mean= 3.4). There are currently over 30 breeding pairs in Scotland (Evans & Wilson, in prep.). One territory on Mull has been continuously occupied by the same birds since at least 1986 (RSPB unpublished data). In Finland and Sweden, White-tailed Eagle territories may be in use for decades or even centuries (Helander, 1988, Stjernberg, 1995), even though almost all nests in these countries are in trees. Territories there were often re-occupied following absences of 20 to 40 years (Stjernberg, 2003). Several of the territories currently occupied on Skye are traditional territories occupied by the original population (MacLennan, pers. comm.).

Distances between alternative nest sites

Alternate nests on Rum were usually 'a mile or two' from each other (Love, 1988). On Mull, a pair which first bred in 1997 moved several kilometres to build a new nest site in 1998, built another nest close to this in 1999, and bred in this same nest in 2000. Alternate nest sites on Skye up to 12km apart have been recorded (MacLennan, pers. comm.).

Distances between alternately used nests have also been measured for a population in Lithuania. The breeding population there was nearly extinct by the mid 19th century, and then increased in the 1980s, with the first breeding records this century occurring in 1987 (Jusys & Mecionis, 1992). Pairs nested between 0-1300m from the previous year's nest (mean= 540m, Jusys & Mecionis, 1992). If nests failed, birds settled at a distance of 200-1300m (mean= 500m, Jusys & Mecionis, 1992). In Israel, the first nest following re-introduction of the White-tailed Eagle in 1996 was in 2000, and an alternative nest was 200m away from this nest (Hatzofe, 2003).

Nearest-neighbour distances

Nearest-neighbour distances vary, with the nearest-neighbour distances in Norway being as short as 1-2km, and in some cases even closer (250-400m), in areas with high densities of breeding eagles (Folkestad, 2003). A nearest-neighbour distance of just1km has also been recorded on Skye (MacLennan, pers. comm.). Mean nearest-neighbour distance in a study by Halley in Norway was about 4.6km (Halley, 1998). Mean nearest-neighbour distances on regions of the Baltic coast in Sweden varied from 4.4km to 13km (Helander, 2003). Mean distances in regions in Lapland range from 9km to 21km with a shortest distance of 3km (Helander, 2003). Shortest nearest-neighbour distances are just 250m in Poland (Mizera, 2003). Nearest-neighbour distances between three nests were 6km and 7km in Latvia, and 9km between another pair of nests (Mecionis & Dementavicius, 2003). Median distance between 14 nest sites in Schleswig-Holstein, Germany, was 6.8km (Struwe-Juhl, 1996a, 1996b). The distance between nests belonging to neighbouring pairs in Lithuania was 3-22km (mean= 12km, Jusys & Mecionis, 1992).

Densities

In Israel, released birds with established territories rarely leave their territory (Hatzofe, 2003). In coastal areas in Norway, where food is plentiful and nest sites at a premium, territories can be 'squeezed' with only a small area around the nest being defended and foraging areas used by more than one pair of birds (Evans, 1999). Densities in Norway in high food areas are around one pair per 12-14km of shore; whilst in Fjord landscapes they are one per 20 to 25km of shore (Folkestad, 2003). In the densest populations, in a study by Halley, there were 16 White-tailed Eagle territories over 760km^2, and densities were higher along the coasts, for example on one large island an area of 760km^2

held 20 pairs, and in Frøya 250km^2 holds about 30 pairs (D. Halley, pers. comm.). In some regions of Germany breeding densities are very high (4-5 pairs per 100km^2) (Hauff, 2003). Over a long period of time, numbers of breeding pairs in Poland have increased, even in areas that are already heavily occupied, seemingly at carrying capacity (Mizera, 2003). For example, within 13km^2 there were 6-7 breeding pairs and 22 nests (Helander & Mizera, 1997). In Lithuania 9 pairs of White-tailed Eagles had densities of 3 pairs per 100km^2 (Mecionis & Dementavicius, 2003). The highest density of White-tailed Eagles in the Ukraine is 0.6 pairs per 1000km^2 (Gavrilyulk & Grishchenko, 2003).

Territory sizes

Mean territory size in a study area in Russia is 17km^2 as oppose to 5.5 km^2 for Steller's sea eagle (Masterov, Runde conference 1996). This is equivalent to a circle of radius 2.3km. In 1995, the population in Schleswig-Holstein, Germany increased to 16 breeding pairs, with mean territory size established being 62km^2 (standard deviation= 35km^2, n= 8, Struwe-Juhl, 1996a, 1996b). This is equivalent to a circle of radius 4.4km. These radii will be crude approximations, as territories will not be circular. Most foraging activity in Schleswig-Holstein, Germany, took place within 5km of the nest site but some pairs flew up to 13km to forage (Struwe-Juhl, 1996a, 1996b). In a population of White-tailed Eagles in Russia, birds hunted a mean area of 13.1km^2 (equivalent to a circle of radius 2km), and covered an area as far as 2.5-3km from the nest (Masterov, 2003).

Mapping White-Tailed Eagles

All home ranges occupied since the re-introduction programme began (except for one: grid reference not currently available) have been mapped at the 1km square level. Nest locations have frequently not been supplied, but those that have been will also be mapped. All of these will be included, irrespective of whether they have been used recently, given the fact that the population is small and localised although expanding in numbers and range, and historical nest sites are often re-used.

Productivity of White-tailed Eagles is reduced within 1km of wind turbines. White-tailed Eagles appear to be highly susceptible to collision with turbines, or with associated power lines. Given the small size of the population, collision with wind turbines may have a population level affect on the White-tailed Eagle in Scotland. There is not a great deal of information on foraging ranges, and none from Scotland, but White-tailed Eagles elsewhere generally seem to range between 2km and 5km from the nest (and up to 13km).

White-tailed Eagle home ranges (1km squares) and nest locations (6-figure grid-references) were buffered by a radius of 5km, and this area classified as 'high sensitivity'. This precautionary distance is considered reasonable based on the extremely small and localised nature of the population.

Locations of all sightings of White-tailed Eagles since the beginning of the re-introduction programme began were plotted, and squares classified by how many sightings they contained, in order to investigate whether this flagged up any frequently used areas. Most areas with a particularly high use fell within the 5km buffers so it was decided not to add sightings data to the map.

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8 Hen Harrier

Introduction

The Hen Harrier *Circus cyaneus* has a widespread global distribution, occurring in both North and South America as well as through temperate and boreal regions of the Palaearctic, from the Atlantic coasts of Western Europe to Kamchatka and Sakhalia (Potts, 1998). The species is polytypic, with the nominate race *C. c. cyaneus* occurring throughout the Palaearctic, and the other two other races occurring in North and South America respectively. Global population trends have not been quantified, but there is evidence of a population decline (Ferguson-Lees *et al*., 2001). The Hen Harrier is widely but patchily distributed across much of northern and central Europe, with the European breeding population numbering between 8 332 and 10 840 pairs (Hagemeijer & Blair, 1997). The European breeding population underwent a large decline between 1970 and 1990 (BirdLife International, 2004). These declines abated to some extent during 1990 to 2000, with many populations stabilising, and the species declined only slightly overall (BirdLife International, 2004). Nevertheless, the population remains far below the level preceding the declines (BirdLife International, 2004).

Hen Harriers were once common and widespread throughout the UK, but became virtually extinct in mainland Britain by the start of the 20th century, due mainly to persecution by gamekeepers (Watson, 1977). At this stage, the remaining population was largely confined to Orkney and the Outer Hebrides. It then increased around the middle of the century, due to a reduction in shooting interests, and protective legislation, which was introduced in 1954. Upland afforestation between the 1940s and 1970s favoured Hen Harrier recolonisation of mainland Britain, at least in the early stages of development, and the population increased to an estimated 500 breeding pairs with a further 250-300 pairs on the island of Ireland by the mid 1970s (Watson, 1977).

The UK and Isle of Man breeding population was estimated at 578-700 pairs in 1988-9 (Bibby & Etheridge, 1993), and there were an estimated 578 territorial pairs in the UK and Isle of Man in 1998 (Sim *et al*., 1999). Numbers in mainland Scotland and Wales remained virtually unchanged between the two surveys, but a decrease of about 50% occurred in England and Orkney (Meek *et al*., 1998, Sim *et al*., 2001). These declines were linked to habitat loss and deterioration, and persecution (Etheridge *et al*., 1997, Scottish Raptor Study Groups, 1998, Holmes *et al*, 2000). The 2002 national survey showed an increase of 44% for the UK breeding population to 749 pairs, and an increase of 45% from 436 to 633 breeding pairs in Scotland (Sim *et al*., in prep.).

Despite this recent increase, there are still concerns over the population status of the Hen Harrier due to heavy illegal persecution, which is on a sufficient scale to limit its numbers and range in the UK (Etheridge *et al*., 1997, Potts, 1998). The Hen Harrier favours flat, upland areas to breed on, also suitable locations for wind farms, and is potentially at risk from both collision with wind turbines and also disturbance effects.

The Hen Harrier is a species of high conservation concern in the UK (Gregory *et al*., 2002) and is listed onAnnex 1 of the EU Birds Directive. Two hundred and twenty nine pairs, or 47% of the British population, occur within Special Protection Areas (Stroud *et al*., 2001).

Home Ranges

Site fidelity

In the absence of illegal persecution, breeding Hen Harriers would be relatively sedentary. Having bred once, wing-tagged harriers usually nest in the same area in successive years. The median distance moved between breeding sites in Scotland in successive years found by Etheridge *et al*. (1997)

was 0.71km (n= 51). Picozzi (1984) found that on Orkney females which moved to a new territory moved further following a failure (2.29km +/- 2.41 S.D., n= 73) than following success (1.32 km +/- 0.90 S.D., n= 44), however Etheridge *et al.* (1997) found no such effect.

Range size

Harriers are not known to actively defend hunting territories and hunting ranges can overlap considerably (Watson, 1977, Redpath, 1991, Arroyo *et al*., 2005), which led Watson (1977) to suggest that males will only defend a nest territory of 600m in diameter. However, interactions between neighbouring birds have also been observed (Picozzi, 1984, Garcia & Arroyo, 2002) probably leading to avoidance of occupied areas.

Habitat close to the nest is used disproportionately when hunting during the breeding season (Picozzi, 1978, Madders, 2003, Thirgood *et al*., 2003), and this has also been shown during the pre-settlement period (Amar & Redpath, 2005). A number of estimates of the size of hunting territories have been made and range size appears to vary considerably between regions. This is likely to be a function of prey supply and the availability of nesting and foraging habitat (Madders, 2004). Many of these studies also show that the home range increases as the nesting period progresses and that males have a greater hunting range than females (Madders, 2004).

There is little quantitative information on the hunting ranges of Hen Harriers in Scotland. Picozzi (1978) made visual observations of Hen Harriers in Scotland and suggested that most hunting by females was carried out within sight of the nest until young were feathered. It was estimated that ranges were greater than 15km^2 in area, equivalent to a circle with a radius over 2.2km. Thirgood, Redpath & Graham (2003) found a significant positive relationship between females hunting and nest site proximity at Langholm, although the relationship was less significant for males.

Similarly, Martin's (1987) study of radio-tracked breeding Northern Harriers *Circus hudsonius* (a very close relative of the Hen Harrier) in North America found that female harriers never ranged further than 2km from their nest sites, whereas males spent 26% of their time ranging over 2km from the nest. They estimated ranges of 15.7km^2 and 1.13km^2 for males and females respectively, equivalent to circles with radii of 2.3 and 0.3km. Another North American study estimated range sizes at 72-366km^2 (approximated by radii of 2.6-6.1km, Thompson-Hanson, 1984). Using visual observations in continental Europe, Schipper (1973) estimated average range sizes of 5km^2 and 1.4km^2 for males and females, respectively (equivalent to circles with radii of 1.3km and 0.7km respectively). More recently, radio-telemetry studies at two Scottish sites, on Orkney and at Langholm, have estimated average ranges of 9.3km^2 for males and 4.6km^2 for females (equivalent to circles of radii 1.7km and 1.2km respectively, Arroyo *et al.* 2005). Furthermore, it has been shown at Langholm that the amount of certain habitats within a 2km circular radius of a nest influences the type of prey brought to that nest (Amar *et al*., 2004). Watson (1977) considered that in Scotland male birds will hunt more than 3km from the nest, and are often recorded 4km or more from the nest site. Watson also concluded that female Hen Harriers can regularly hunt 2km to 3km from the nest, and some have been recorded foraging as far as 8km from the nest. Hunting territories are not circular and so radii estimated from ranging areas and give a fairly crude illustration of the extent of a hunting range.

Disturbance, Collision Risk and Buffer Zones

Disturbance

A distance of 250m was used as the search resolution for the 1998 national survey of the breeding Hen Harriers in areas of 'restricted visibility' (Sim et al., 2001). Disturbance free zones of 500-600m (Petty, 1998), and 500-1000m (Currie & Elliot, 1997) have been recommended around occupied Hen Harrier nests in advice for forestry workers in Britain, although it is not clear what these distances were based on.

Mike Madders, based on his knowledge of Hen Harriers' tolerance to a wide cross section of disturbance events, estimated that nesting and foraging harriers would be displaced from distances of around 500m from each construction work site, with some disruption to nesting and foraging behaviours extending up to 1km along lines of sight (Madders, 2004). Estimates for disturbance distances from turbines were 250m, with some disruption to nesting and foraging behaviours extending up to 500m (Madders, 2004).

In a more recent review of the impacts of wind farms on Hen Harriers, Whitfield and Madders (2005) looked at eight studies of displacement effects by wind farms in the USA and continental Europe and found that only one study found good evidence of displacement. This study, by Johnson et al. (2000) found evidence of both small scale (<100m from turbines) and larger scale avoidance of turbines by harriers in the year following wind farm operation. Whitfield and Madders (2005) concluded that if displacement of foraging occurs it is likely to be limited to within 100m of turbines. However, it is possible that the unusually high levels of persecution that Hen Harriers in the UK are subjected to would make them more susceptible to disturbance.

There are fewer studies of displacement impacts on nest site selection, however, but preliminary results from Argyll in Scotland and Northern Ireland indicate that local displacement of nesting attempts may occur within 200-300m of turbines (Whitfield & Madders, 2005).

The review concluded that the main likely impact of wind farms on harriers when situated away from nest sites is thus likely to be collision with turbines. Disturbance could have an effect on the productivity of nests close to wind farms, however, and these effects were not discussed in the review. Productivity could be reduced by disturbance via a number of mechanisms, and not just nest failure; for example, a study of Marsh Harriers in Spain found that human disturbance reduced breeding success by reducing provisioning rates leading to decreased fledgling survival (Fernandez & Akzona, 1993).

Collision Risk

Hen Harriers typically hunt low over the ground (Watson, 1977) but may be more at risk of collision with turbines during display flights or flights made as newly fledged birds (Madders, 2004). Display flights by male harriers occur mainly within 500m of the nest, but can be made up to 1km away (Madders, 2004). Thus, collision risk for displaying birds is likely to be greatest about 500m from the nest, declining to there being little risk at a distance of 1km or so away (Madders, 2004). There is no direct evidence of birds colliding with obstacles during aerial displays, probably because the likelihood of observing it is very low. However, in Argyll, west Scotland, at least two male Hen Harriers were killed due to collision with overhead lines close to traditional nesting areas in late April/early May, the peak display period (Madders, 2004). Juveniles are regularly killed in their first few weeks due to collision with fences (Madders, 2004).

It is not known how harriers respond as they approach operational turbines but if they respond by actively gaining height then this would substantially increase collision risk. This risk would potentially affect harriers throughout the year and would apply to breeding birds several kilometres from their nests (Madders, 2004).

Whitfield & Madders (2005) reviewed nine studies of collision rates at wind farms where Hen Harriers occur; mortalities of Hen Harriers were recorded in three studies. Two of these involved migrating birds, and the third involved breeding and passage birds. Only one of the studies recorded mortality of more than one bird, this was at the Altamont Wind Resource Area, California, where three casualties occurred over five years. The review concluded that mortality rates are probably only rarely expected to be high, and this may be when turbines are located in the vicinity of several nest sites.

Mapping Hen Harriers

Hen Harrier locations from the 2004 and 1998 national surveys were plotted. Although these were stratified random surveys and coverage was not complete (except in Orkney), areas outside the official survey squares were also surveyed, such that the 2004 national survey located 72%, and the 1998 national survey 77%, of the estimated Scottish population (Sim *et al*., 2001, Sim *et al*., in prep.).

Survey data does not always give nest locations; some grid references refer just to sightings of hunting or displaying birds. Given that estimates of hunting range radii range from around 1km to 2km, that it is desirable to buffer nests by a distance of 1km, and also that nest locations may be moved between years by up to about 1km, a buffer of 2km around each 2004 grid reference has been applied and this area classified as 'high sensitivity'.

The area within 2km of 1998 national survey grid references has been classified as 'medium sensitivity'. This was so that recently, but not currently, occupied locations were included. This will include areas where birds have been lost due to persecution, which otherwise could have been recolonised (or could be recolonised in the future).

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These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in Scotland with respect to a suite of sensitive bird species. Thanks to Arjun Amar, Rhys Bullman, Sam Gardner, Rowena Langston, James Pearce-Higgins, and Innes Sim who provided helpful comments on an earlier draft of these sensitivity criteria.

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9 Golden Eagle

Introduction

The Golden Eagle has a northern global breeding distribution that includes the Palaearctic (as far south as northern Africa and south-east Asia), as well as North America. In the UK, the species occurs primarily in the uplands, although there are several pairs that exploit coastal habitats. The vast majority of the population is found in the Scottish Highlands (Dennis *et al*., 1984, Green, 1996, Watson, 1997). The global population is estimated at 50 000 to 100 000 pairs (Watson, 1997), with the European population standing at 5239 to 5616 pairs (Hagemeijer & Blair, 1997). Spain holds the largest of the European populations, followed by Norway, Sweden and the UK. Global population trends have not been quantified, but populations appear to be stable (Ferguson-Lees *et al*., 2001) with the exception of some regions such as the east Baltic and south east Asia where there have been declines (Stroud *et al*., 2001). The European breeding population was stable between 1970 and 1990, and remained stable or increased in the majority of countries between 1990 and 2000 (BirdLife International, 2004). However, because the small size of the European population makes it vulnerable, *Birds in Europe* evaluates the Golden Eagle as 'Rare'.

Prior to 1800, there were possibly 500 pairs in Scotland and at least 50 in England and Wales (Brown, 1976, Holloway, 1996). The UK population declined as a result of persecution in the 18th and 19th centuries to a low point of only 80 regularly used eyries by 1870 (Watson, 1997). The British population is thought to have increased since the 1950s following the introduction of protective legislation, notably the Protection of Wild Birds Act in 1954 (Watson, 1997). The last national survey in 2003 found 430 pairs of Golden Eagles breeding in Scotland (and one in England, Eaton *et al*., in prep.), in comparison with about 422 pairs in the 1992 national survey (Green *et al*., 1996). Densities are highest in the western Highlands and Islands.

The UK Golden Eagle population constitutes around 7.4% of the European breeding population, and virtually all of these birds occur in Scotland. The Golden Eagle is a species of medium conservation concern in the UK (Gregory *et al*., 2002) and is on Annex 1 of the EU Birds Directive. Around 60 pairs, or 15% of the British breeding population, occur with Special Protection Areas (Stroud *et al*., 2001).

Prediction of Golden Eagle Home Ranges

Golden Eagles breed habitually in the same nest sites, though pairs use alternate nest sites in different years up to 6km apart (Watson & Rothery, 1985).

A simple model to predict Golden Eagle's home ranges was developed by McGrady *et al*. (1997, 2002), by radio-tracking birds in Argyll. This is known as the RIN (Research and Information Note), RIN 292, or McGrady model and involves a number of stages:

1 *Range Centre*

The range centre is found by taking the mean nesting location over the past 10 years. If there are clusters of nests that have means over 2km apart then there will be two centres in the range.

2 *Core Area*

This is the central core of the range, which encompassed 50% of the ranging locations of eagles. In its simplest form, it is a circle of 2-3km radius around the range centre.

3 *Territory boundary*

a Where neighbours were present, this is defined by drawing a line between the neighbouring range centres, and then drawing a line perpendicular to this line at the mid-point between the two centres. This is done for all of the neighbouring ranges, and the lines joined to define the territory (a Thiessen polygon).

b In the absence of near neighbours, eagles ranged up to 9km from the centre (although 98% of records were within 6km of the range centre). A curved line is drawn at 6km from the centre to connect adjacent boundary lines drawn in step 3a.

4 *Altitudinal cut-off*

Eagles preferred an altitude range of 150-500m (except on Mull where it was sometimes lower). High areas were avoided, but this may not be true in the Eastern Highlands where Ptarmigan and mountain hare are more plentiful. Thus, different cut-offs were used for eagles nesting at different altitudes:

- High altitude ranges, e.g. the Cairngorms, the altitudinal cut-off outside the core area was defined as 150-200m above the valley floor.
- Medium altitude ranges, e.g. mainland Argyll, the altitudinal cut-off was defined as 150m out of the core area.
- Low altitudes ranges, e.g. Isle of Mull, all altitudes were used, excluding areas of high human activity.

This model was refined by McLeod et al. (2002) to create the PAT (Predicting Aquila Territories) model. The range centre was estimated as before, but a more complex model predicting percentage use of the range at a 50m x 50m pixel level was created. Whilst the RIN model used a set cut-off distance of 6km to define the territory boundaries in the absence of near neighbours, the PAT model allows for the fact that this may vary according to density of breeding eagles, for example it is probably less in high density areas such as some Hebridean Islands. Maximum ranging distance was found to correlate with the area of the Thiessen polygon for a range (see 3b above). Areas close to the centre of a range were assumed to be used more. The maximum ranging distance was used to define the percentage use in concentric rings around the range centre, such that the total use was 100%. Preference for ridge features (to aid soaring) was incorporated, as was avoidance of unsuitable habitat: fresh water bodies, sea, woodland over 12 years old and areas of high human activity (with various buffer zones, see below). When compared to the RIN model, the RIN predicted larger ranges.

Disturbance

There is little information on disturbance distances, or what activities are affected most by disturbance (McLeod *et al*, 2002). The effects of disturbance will also vary throughout the range, in recent years an expanding population has moved into areas which are more disturbed to establish new breeding ranges (Haller, 1996). Such a situation does not apply to Scotland (Watson & Dennis, 1992).

Golden Eagles are most sensitive to disturbance near the nest in February to July (McGrady *et al.*, 1997), and disturbance can influence Golden Eagle behaviour and productivity (Watson, 1997). Disturbance free distances of 900-1000m radius around the nest have been recommended in Britain (Petty, 1998). This may vary, being increased to around 1.5km if disturbance is directly in the sight line of the nest, and decreased if the disturbance is concealed in some way (to a minimum of 750m, McGrady *et al.*, 1997). Distances of 200-1600m have been recommended in North America (Suter & Joness, 1981).

Bergo (1984) found that all nests (in the study area in Norway) were found at a distance of more than 500m from permanent human settlement, and usually more than 1km away from roads. Three nests were found less than 500m away from a cabin or mountain farm house (minimum 250m), but one occupied nest situated 450m from a cabin was deserted during the incubation period.

McLeod *et al.* (2002) assumed areas of human activity were avoided by eagles, and used the following buffer zones:

Single building	250m
Cluster	400m
Village	600m
Town	800m
Single carriageway roads	300m
Dual carriageway roads	500m.

These were base on limited observations.

Two studies in the USA found no evidence of disturbance displacement of Golden Eagles due to wind farms (Johnson *et al*., 2000, Schmidt *et al*, 2003), but a third study at the Altamont Pass wind farm in California found some evidence of displacement in comparison with a control area (Hunt *et al*., 1995). Golden Eagle behaviour was monitored before and after construction of the Beinn an Tuirc wind farm in Argyll in 2001 within the range of a pair of Golden Eagles. The eagles' range size remained the same, but the eagles appeared to change their ranging behaviour to avoid the wind farm site (although there are problems with interpreting this result as the construction of the wind farm coincided with some local land management changes, Walker *et al*., 2005).

Collision Risk

Wind farms located within Golden Eagle ranges can cause deaths due to collisions (Hunt, 2002). At the Altamont Pass Wind Resource Area in California, there were an estimated minimum of 39 Golden Eagle mortalities due to collision, with proportionally more Golden Eagles being killed than would be predicted by chance. This is likely to have been due to hunting behaviour (Orloff & Flannery, 1992). A review of the European literature found one reported casualty of a Golden Eagle due to collision with a wind turbine in Spain (Hötker *et al*., 2004). See also White-tailed Eagle sensitivity criteria.

Golden Eagle Mapping

Data from the 1992 and 2003 National Surveys were used in attempt to map Golden Eagle nest locations in Scotland as fully as possible. Some of the data from these surveys is in the form of range centres, but most are nest locations.

These range centres/nest locations were then buffered by circles of radii 2.5km in order to map core ranges. This is based on the RIN model which recommends a distance of 2-3km. McLeod *et al*. (2002) and Whitfield *et al*. (2001) refer to the RIN model as giving core areas with radius 2.5km. Core areas for records from 1992 were classed as 'medium sensitivity' and those for records from 2003 were classed as 'high sensitivity'. This is because it was considered difficult to justify a classification of 'high sensitivity' for a square where no bird was currently present, but it was considered important to include squares containing nest sites which may not be used currently (in some cases due to persecution), but could be used again in the future.

The core ranges (circle radius 2.5km round range centre) represent where an eagle will be 50% of the time. In order to account for where an eagle will be the rest of the time, concentric rings with inner radii of 2.5km and outer radii of 6km were drawn round the range centres/nest locations, and the area between the 2.5km radii circles and 6km radii circles classed as 'medium' sensitivity. This is after the RIN model, which found 98% of records to be within 6km of the range centre (McGrady *et al*., 1997, 2002).

Golden Eagles are within home range core areas for 50% of the time, and avoidance of these will reduce risk of collision by territorial adults as well as minimising the impact of indirect habitat loss. It was decided to use core areas rather than the smaller recommended disturbance free areas, as it has been shown that if as little as 10% of a Golden Eagle's core range is afforested it can lead to a large reduction in productivity (Whitfield *et al*., 2001), and that home ranges may be abandoned if up to 40% of the core is occupied by closed canopy forestry (Watson, 1992). Whitfield *et al.* (2001) used the more sophisticated PAT model to estimate core range, which was the area in which an eagle spent 80% of its time, but this was roughly equivalent in area to the RIN model version found by buffering range centres by 2.5km (and in which an eagle spends about 50% of the time).

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10 Peregrine Falcon

Introduction

The Peregrine Falcon has a wide global distribution, being absent as a breeder from only a few regions of the world (Cramp & Simmons, 1980, Ratcliffe, 1993). Likewise, Peregrines occur widely throughout Europe, although they are generally highly dispersed and nest at low densities. Organochlorine pesticide contamination caused several population crashes throughout Europe in the 1950s and 1960s, and the Peregrine has since been the subject of intensive conservation efforts worldwide (Crick & Ratcliffe, 1995). In Britain, the population reached its lowest point in 1963 when only about 360 pairs bred (Ratcliffe, 1972).

National surveys of the UK Peregrine population have been conducted every ten years since 1961 (Crick & Ratcliffe, 1995). The 1991 survey revealed that the Peregrine population was at its highest recorded level since records began, with the population having grown in both breeding density and range (Crick & Ratcliffe, 1995). This was due to an increase in productivity and probably also survival, reflecting the disappearance of pesticide problems, reduced persecution through protection efforts, and locally enhanced food supply through the growth of pigeon racing (Crick & Ratcliffe, 1995). Since 1991, the number of occupied territories in the UK and Isle of Man had risen a further 10% to 1 402, compared with 1 283 in 1991 and 874 in the 1930s (before the population crash, BTO web-site). The Peregrine is found at low densities thoughout much of the UK (except much of central and eastern England), particularly in coastal and upland areas. Approximately 95% of the Peregrine's potential UK breeding range is now occupied (Newton, 1994) although there has also been some limited spread into a few urban areas.

However, despite these recent increases, regional trends have varied, with the 2002 national survey showing increases in most parts of England, southern Wales and southern Scotland, but declines in north and west Scotland, north Wales and Northern Ireland (BTO web-site). The 1991 survey showed that in the north and west Highlands, the Hebrides and the Northern Isles, the species remained below pre-1940s numbers. Declines in these populations, and in some other areas of Scotland, in the last ten years are probably due to illegal killing, the continuing impacts of environmental pollutants, and habitat deterioration through overgrazing (Scottish Raptor Study Groups, 1998, DETR/JNCC Raptor Working Group, 2000).

The Peregrine is listed under Annex 1 of the EU Birds Directive and is a species of medium conservation concern in the UK (Gregory *et al*., 2002), which currently holds about 20% of the European breeding population (Anon., 2000). The UK SPA suite currently holds 109 pairs, or 9% of the British breeding population, in ten sites, five of which are in Scotland (Stroud *et al*., 2001).

Home Ranges etc

Philopatry

Laying occurs late March to early May (Ratcliffe, 1993). Re-nesting may occur following failure of the first brood up to mid August (Ratcliffe, 1993). Ratcliffe (1993) states that the majority of females return to the same place where they nested the previous year, and limited evidence from re-trapping of marked birds by Mearns and Newton (1984) suggests that males have the same tendency. However, alternative nest ledges and cliffs may be used. For example, one Lakeland pair has been found to move 12km for a repeat laying (Ratcliffe, 1993). This distance is exceptional, although in several inland areas, distances of 5-6.5km between alternative nest sites in one territory have been observed, but the average is just 2.7km in the southern uplands (n= 28 territories); 2.4km in northern England (n= 40) and 2.1km in north Wales (n= 24) (Ratcliffe, 1993).

Hunting Ranges and Territory Sizes

Weir (1978) in his analysis of grouse predation by Peregrine Falcons assumed that most grouse taken would be within 2km of the nest. Weir (1977, 1978) has given reasons for concluding that central Highland Peregrines took about 70% of their prey within 2km of the nest. Peregrines can hunt further from the nest than this on occasion, with distances of 6km (Weir, 1977, 1978) 13km (Mead, 1969), 16km (Ratcliffe, 1980), and 15km (Glutz *et al*., 1971) being noted.

Average nesting territory sizes of 63km^2 have been reported for south west Scotland, 66km^2 for the south and east Highlands, 90km^2 for the central Highlands, 192km^2 for the north and west Highlands, and 70km^2 for the north and east Highlands. These areas are equivalent to circles with radii of 4.8km, 4.6km, 5.4km, 7.8km and 4.7km respectively.

Home ranges may overlap considerably. Cade looked at Alaskan Peregrines and came to regard the nesting cliff as the centre of a large home range including lots of hunting ground not defended in the usual territorial sense. Nearly all birds defended a central core of 100m, and could defend an area up to 1.6km from the nest (Cade, *in litt*.). Ratcliffe (1993) suggests this may be equally applicable in Britain, but suggests that other types of defence than fighting occur explaining the fact that the usual minimum nearest-neighbour distance is larger than this.

The speed of a Peregrine in level flight is around 100km per hour (Ratcliffe, 1993).

Nearest-neighbour distances

Average nearest-neighbour distances of 5.3km and 5.4km have been reported for south west Scotland, 6.1 and 6.4km for the south and east Highlands, 7.2km for the central Highlands, 6.2km for the north west and north east Highlands, 9.0km for the north and west Highlands, 5.8km for the north and east Highlands, 5.8km for Orkney and 7.4km for Shetland (Ratcliffe, 1980).

Collision Risk, Disturbance etc

Disturbance

Occupation of nest sites by Peregrines in the eastern United States has been found to relate to both height of cliff and remoteness from human presence, with very large sheer cliffs never being deserted no matter how much destruction of adults or nest robbery took place (Hickey, *in litt.*). Ratcliffe (1980) suggests that this also seems to apply to Britain, with large amounts of casual disturbance being tolerated at high, inaccessible cliffs. For example, cliffs in Kent and Sussex were occupied up until the war despite people constantly passing along the top or the bottom of them (Ratcliffe, 1980). Up to the mid fifties, birds bred regularly only 70-100m above an extremely busy road in Wales, on a very sheer face (Ratcliffe, 1980). In 1979, one pair nested in the disused part of a still worked quarry, and another on a face immediately above the site of huge excavations with heavy plant in progress right through the nesting season, both were successful (Ratcliffe, 1980). The occasional use of buildings in large cities as nesting places is further evidence of the species tolerance of human presence, and this is increasing rapidly, perhaps indicating some degree of adjustment to human disturbance (Ratcliffe, 1993).

Buffer Zones

Disturbance-free zones for forestry workers of 400-600m (Petty, 1998), and 600-1000m (Currie & Elliot, 1997) have been recommended around occupied Peregrine nests in Great Britain. In Poland, a strict protection zone of 200m and a seasonal protection zone of 500m is enforced (http://free.ngo.pl/eagle/ochrona.htm).

Collision Risk

A Peregrine was amongst the victims of 55 birds to be killed by 23 operational wind turbines in the course of a year in Zeebrugge, Belgium (Everaert *et al*., 2002). After correction for biases it was estimated at least 531 birds had collided in total (although most of these were gulls, Everaert *et al*., 2002). A review of European literature found two recorded Peregrine fatalities as a result of collision with wind turbines, both of these in Belgium (Hötker *et al*., 2004). There appears to be a low risk of collision with wind turbines for Peregrine. However, the speed attained by hunting Peregrines and the possibility of focus on the pursuit rather than the bird's surroundings (e.g. Madders & Whitfield, 2006) must predispose them to collision risk.

Mapping Peregrine Falcons

Coverage of the 2002 national survey in the United Kingdom was about 93%.

Given that 70% of ranging behaviour is expected to occur within 2km of the nest, nest locations were buffered by 2km and this area classified as 'medium sensitivity'. This is to include the main part of the foraging range of nests found within the 2002 survey, and to highlight areas that are also likely to contain alternative nest sites. Peregrine have not been included as 'high sensitivity' due to their widespread distribution.

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11 Black Grouse

Introduction

The Black Grouse is distributed throughout continental Europe and Siberia. During the last century, Black Grouse populations have declined in numbers in most parts of western and central Europe, and in some places gone extinct due to habitat loss, degradation and fragmentation, caused by changes in land use, particularly agricultural intensification (Cramp & Simmons, 1980, Baines, 1994). The British Black Grouse population has decreased considerably over the last 150 years, with a decline in numbers of Black Grouse shot in Scotland and northern England of around 90% since 1900 (shooting bag counts reflecting population densities; Baines & Hudson, 1995). The Black Grouse's British range has also severely contracted over the last century, with Black Grouse previously common throughout the southern counties of England, but now largely confined to Scotland and north east England, with a small population in Wales (Gibbons et al., 1993, Baines & Hudson, 1995). This range contraction has continued in recent decades, with Black Grouse being found in 28% fewer 10km squares between the 1968-71 and 1988-91 breeding bird atlases (Sharrock, 1976, Gibbons et al., 1993). Range contraction is most severe from the south and west of both the British and the Scottish range (Baines & Hudson, 1995).

The second national survey of Black Grouse took place in 2005, and gave a population estimate of 5 078 lekking males in Britain; a decline of 22% since the first national survey in 1995/6 (Hancock et al., 1999, Sim et al., in prep.), and 80% since the previous population estimate of 25 000 in 1990 (Baines & Hudson, 1995). Numbers in Scotland fell by 29% between the 1995/6 and 2005 national surveys (Sim et al., in prep.). The magnitude of decline varied regionally, being largest in the southeast and southwest, with smaller declines in the north and east of Scotland (Sim et al., in prep.). Scotland was estimated to hold about 66% of the national population in 2005 (Sim et al., in prep.). Concern over the decline in range and numbers of the British Black Grouse population has led to several conservation initiatives (Calladine et al., 2002). The Black Grouse is a species of high conservation concern in the UK (Gregory et al., 2002) and is protected under Annex II/2 of the EU Birds Directive.

Dispersal, Home ranges etc

Mating system

Black Grouse males are polygamous and play no part in rearing young. They display communally and mate on traditional arenas (leks) where they defend a small area. Males attend leks through the year except when moulting but the main period of activity is between March and June and, in some areas, in late September and October (Koivisto, 1965, Hjorth, 1968). In Scotland, females visit leks from April onwards (Picozzi *pers. comm.*, in Cayford, 1993). Surveys of Black Grouse are usually based on counts of males at lek sites as leks are relatively easy to find and view compared to nests.

Several females may nest in the vicinity of the lek, and clutches are laid between late April and early June. There is usually one brood. Most females will nest within 1.5km of the Black Grouse lek, so management for Black Grouse is targeted at this area (Anon., 2003). Willebrand (1988) found that females usually nest within a 2km radius of the lek. Both males and females remain in the same general locality throughout the year and rarely move more than a few kilometres from the lek they attend in spring (Warren & Baines, 2004). Warren and Baines (2004) state that in good, continuous habitat, leks are approximately 2km apart, and thus most birds attending the leks are found within 1km of the lek (Warren & Baines, 2004). They suggest that suitable habitat needs to be available within an area of 200-500ha around a lek (equivalent to circles of radii 0.8-1.3km, Warren & Baines, 2004).

Dispersal and site fidelity

Dispersal in Black Grouse is confined to first-year hens, with adults of both sexes and first-year cocks showing high site fidelity (Warren & Baines, 2002, Caizergues & Ellison, 2002). Male Black Grouse show a high degree of natal philopatry. In a study in the North Pennines, natal dispersal resulted in none of the eight first-year hens remaining within their study area, whilst all 11 cocks remained, all within 1km of their natal site (Warren & Baines, 2002). Birds of both sexes sometimes occupy separate winter and summer ranges and generally remain faithful to the ranges they establish in their first winter and summer in subsequent years (Willebrand, 1988, Caizergues & Ellison, 2002). In the French Alps, ten birds (six males and four females) captured as juveniles, and two males and one female captured as yearlings, were monitored during at least two consecutive summers by radio-tracking (Caizergues & Ellison, 2002). The summer ranges of 92% (12/13) of 2-year old birds overlapped with the summer ranges occupied as yearlings, with the exception being a female who moved her nest site by about 1km. Ninety one percent of males marked as juveniles were still in the study area the next summer. Furthermore, two one-year-old cocks occupied about the same summer range that they had established as yearlings (Caizergues & Ellison, 2002).

Densities

The largest lek recorded during the 1995/6 national survey had 17 males. Densities varied from 0 to 2.4 displaying males per km^2 of apparently suitable habitat (Hancock *et al.*, 1999). In a survey of 69 leks in Perthshire, the numbers of males at a lek ranged from 2-30 (median and mode 7; Robinson *et al.*, 1993). Thirteen percent of leks had 15 or more cocks and 25 single displaying males were found. Densities ranged from 0.8-2.3 males per km^2, with a mean of 1.6 males per km^2. This mean is similar to estimates from other British studies, and is within the range of densities found by Baines (1992) but is slightly higher than that found by Picozzi (unpublished) for parts of Deeside (Robinson *et al.*, 1993). Higher densities have also been reported by studies in Sweden, the Alps and Estonia, but these studies have either used smaller areas within which birds may be concentrated, or only favourable habitats have been searched.

Adult home ranges

Male home ranges tend to be larger than those of females. Johnstone (1969) found that home ranges of lekking males in north east Scotland were about 500ha (equivalent to a circle of radius 1.3km). Robel (1969) reported home ranges of males in Scotland of 303-689 ha with a mean of 458 +/- 126 S.D. (equivalent to a circle of 1.2km), and Picozzi (1986) obtained ranges of 355ha +/- 178 S.D. for males (equivalent to a circle of radius 1.1km), 86ha +/- 37 S.D. for moorland females (equivalent to a circle of radius 0.5km) and 45ha +/- 28 S.D. (equivalent to a circle of radius 0.4km) for woodland females in Scotland.

In coniferous plantations in Wales, home ranges of radio-tagged males averaged 325ha +/- 71 S.D. (equivalent to a circle of radius 1km, Cayford, 1990), whilst mean home range of two females was 50.2ha (equivalent to a circle of radius 0.4km, Cayford & Hope Jones, 1989). Home ranges of males attending the same lek overlapped considerably and were discrete from males attending adjacent leks. Males did not appear to move between leks. These data confirm that Black Grouse are largely sedentary and that an area of 500-700 ha of suitable forest and moorland may be enough to sustain one lek (equivalent to circles of radii 1.3km to 1.5km, Cayford, 1993). Solitary display sites may be used for just one or two years compared to the regular use of traditional leks (Cayford, 1993).

Brood home ranges

Eggs hatch synchronously, usually in mid June, and young are led from the nest within 24 hours of hatching. Broods travel successively further from the nest site during the first 2-3 weeks before settling into home ranges of 30-60 ha (Wegge *et al.*, 1982). Warren and Baines (2004) describe the home range of Black Grouse broods as usually small, typically 10-30 ha. Starling (1990) also found home ranges in the North Pennines to be small (less than 50ha).

Collision Risk

Powerlines

Collisions with power lines have been found to be a cause of systematic mortality for tetraonids in boreal forest habitats in central Norway (Bevanger, 1995). Clear-felled corridors along 25km of powerlines were patrolled for 850km, and fifty tetraonid collisions were noted, including 14 of Black Grouse. Power line related mortality was higher in winter than summer for tetraonids. Marking of power lines to reduce Black Grouse mortality has been suggested (MacLennan, 2000).

During display flights, Black Grouse usually fly at less than 15m (Hjorth, 1970) but may sometimes reach heights of up to 30m (Koivisto, 1965), which would put them at potential risk of collision with wind turbines (mean lowest sweep of rotor= 27m, n= 84, calculated from SNH data of wind farms in Scotland although this lowest sweep height is likely to be getting higher as new, larger turbines are developed). Langston and Pullan's review (2003) identified Black Grouse as being at risk of both collision with and disturbance from wind turbines (Langston & Pullan, 2003). A review of the European literature found two recorded Black Grouse fatalities as a result of collision with wind turbines in Austria (Hötker *et al*., 2004).

Disturbance

Disturbance of lekking birds has been identified as a severe problem at some isolated sites (Anon., 2003). Protection of important or isolated lek sites from human disturbance is one of the objectives of the species action plan (Anon., 2003).

Mapping Black Grouse

Black Grouse is one of the few non Annex 1 species to be included on the sensitivity map. They have been included due to the rapid rate of population decline, their low dispersive ability, which makes them poor at recolonising their range, and the fact that they are on the top 20 priority species in Scotland list in SNH's species framework.

The 2005 national survey was based on a stratified random sample of 5km squares, constituting approximately 8% of the species' range in northern England and Scotland. Thus, lek counts from national and regional surveys were collated to obtain as full coverage of the Black Grouse's Scottish range as possible. The years from which data were used varied regionally, depending on what was judged necessary to obtain the most complete coverage, but all data were from surveys conducted after 2000.

The Black Grouse Species Action Plan states that most females will nest within 1.5km of the Black Grouse lek, so management for Black Grouse is targeted at this area (Anon., 2003). Likewise, the Forestry Commission targets grants for Black Grouse to the area within 1.5km of important lek sites. A distance of 300-1000m is recommended as a provisional safe working distance from nests and leks for forestry workers in Great Britain (Currie & Elliot, 1997). Estimated home range sizes for Black Grouse in Britain are generally equivalent to the area of circles with radii of around 1-1.5km for males, and 0.5km for females (Johnstone, 1969, Robel, 1969, Picozzi, 1986, Cayford & Hope Jones, 1989, Cayford, 1990, Cayford, 1993). Cayford (1993) suggests that an area of 500-700ha of suitable forest and moorland may be enough to sustain one Black Grouse lek; these areas are equivalent to circles with radii of 1.3km to 1.5km (Cayford, 1993). Warren and Baines (2004) state that most birds attending a lek will be found within 1km of the lek, and that suitable habitat needs to be available within an area of 200-500ha around a lek (equivalent to circles with radii 0.8-1.3km). Thus, it was decided to buffer Black Grouse leks by a distance of 1.5km, and classify the area within this buffer as 'high sensitivity'.

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12 Capercaillie

Introduction

The Capercaillie *Tetrao urogallus* has a global distribution extending from Western Europe to north central Asia (Picozzi *et al*., 1992, Storch, 2001). The nominate sub-species *Tetrao u. urogallus* occurs in Europe from Britain, Scandinavia and central Europe in the east to north west Siberia (Stroud *et al*., 2001). Its European range has decreased and become highly fragmented, owing to forest loss and degradation, particularly in lowland areas (Stroud *et al*., 2001).

Within the UK, the Capercaillie population is restricted to Scotland, with highest densities occurring in east Highland, Grampian and Tayside (Gibbons *et al*., 1993). The original UK population became extinct in the 18th century, possibly as a result of habitat loss and hunting, but was successfully re-introduced in the 1830s (Thom, 1986). The population declined again in the first half of the 20th century due to extensive tree felling, but subsequently recovered, with a peak population size in the 1970s (Eaton *et al*., in press). Since the 1970s, there has been a further decline in numbers and range, with birds being lost from 64% of 10km squares occupied in the 1968-72 breeding bird atlas by 1988-91 (Sharrock, 1976, Gibbons *et al*., 1993). It has been suggested that climate change may have been one of the major causes of the population decline, as a result of increased chick mortality due to wetter June weather, and lack of food for hens due to delayed warming in April (Moss *et al*., 2000). Other contributory causes are habitat loss (Moss *et al*., 1979), degradation of chick-rearing habitat (Baines *et al*., 1994), predation (Summers *et al*., 2004a), and increased adult mortality due to collisions with deer fences (Catt *et al*., 1994, Baines & Summers, 1997, Baines & Andrew, 2003).

The second national survey of the Scottish population, in 1998/9, estimated a population size of 1073 birds (Wilkinson *et al*., 2002). This was a 51% decline since the previous population estimate in 1992-4, equating to a rate of decline of 13% per year (Wilkinson *et al*., 2002). Models suggested that if the demographic factors behind this decline continued to operate in the same manner, the population would decline to just 40 hens by 2014, which is below the probable Minimum Viable Population (MVP) threshold (Moss, 2001). This high risk of extinction has led to a huge conservation effort in recent years, with ongoing projects such as the EU-funded Nature-LIFE project (Urgent Conservation Management for Scottish Capercaillie) and others embarking on a programme of removal or marking of deer fence, predator management and habitat restoration or creation (Eaton *et al*., in press).

The last British national survey of Capercaillie in the winter of 2003-04 estimated a population of 1980 birds (Eaton *et al*., in press). This was an increase of 84% since the previous estimate from the 1998-9 national survey (Wilkinson *et al*., 2002). This increase was not statistically significant, and could be a result of increased survey effort in the 2003-04 national survey (Eaton *et al*., in press). However, a number of regional surveys suggest a recent increase has occurred in the core areas for Capercaillie, particularly in Strathspey (Eaton *et al*., in press). Large concentrations of birds exist in Badenoch and Strathspey, with 66% of Capercaillie sightings from the last national survey being made in this area, and in the RSPB reserves at Abernethy and Craigmore, where 31% of all sightings occurred (Eaton *et al*., in press).

The Capercaillie is a species of high conservation concern in the UK (Gregory *et al*., 2002) and is listed under Annex 1 of the EU Birds Directive. Stroud *et al.* (2001) estimated that 17% of the British Capercaillie population occurs within the eight Special Protection Areas for this species, although 54% of all Capercaillie sightings in the last national survey were within these SPAs, suggesting the proportion could be considerably higher (Eaton *et al*., in press).

Breeding system and home ranges

Capercaillie prefer older native pinewood with dominant Scots Pine and Blaeberry understorey (Eaton *et al.*, in press). They display at communal leks, which are evenly spaced at roughly 2km intervals in continuous forest (1.9km, Abernethy: Summers *et al.*, 1995; 2.1km, Scotland: Picozzi *et al.*, 1992; 2-2.1km, Scandinavia, Weggen & Rolstad, 1986; 1.9-2.4km, Estonia: Viht, 1991), and are attended by one to 20 males (Moss & Picozzi, 1994). The peak display period is late April, but leks can also be used at other times of the year, particularly in autumn (Petty, 2000). Each cock's daytime territory radiates about 1km from the lek like wedges of a pie, with males moving towards the centre when displaying in spring (Moss & Picozzi, 1994). Most females attend one lek and breed within the territories of males attending the leks, although some range widely and visit more than one lek (Petty, 2000). This spatial distribution of leks led Moss & Picozzi (1994) to conclude that the area of forest needed to support a lek is around 300 to 400 ha (Moss & Picozzi, 1994). However, territories can be quite variable, particularly in some of the smaller forest patches in Scotland.

Mean home range sizes of 180ha for males (n= 7, range= 86-329ha) and 44ha for females (n= 2, range= 18-70ha) have been recorded for a population in central Scotland (Picozzi *et al*., 1996). Home ranges of 294ha and 345ha for two radio-tracked males, and 89ha and 482ha for two females have been recorded at Abernethy (Summers & Proctor, 1996). Range diameters of 4km and 4.6km were recorded for two males, with an average female home range diameter of 2km (n= 9, range= 1-24km, Summers & Proctor, 1996).

Dispersal and site fidelity

Following post-fledging dispersal, Capercaillie in Scotland show a high degree of site fidelity (Petty, 2000). Males and females sometimes have separate ranges in summer to those used from autumn to spring, but seasonal movements appear to be relatively small, with great fidelity shown to winter and summer ranges and lek sites (Petty, 2000).

Females appear to disperse more than males in their first year (Koivisto, 1963, Schroth, 1991, Moss & Picozzi, 1994). There is a lot of individual variation in post-fledging dispersal distances, with a radio-tracking study of first-year birds in Scotland showing that 3 of 22 hens moved over 20km from where they were tagged, whilst none of the seven cocks tagged moved more than 5km from their natal area (moss & Picozzi, 1994). This is typical of distances moved by radio-tagged juveniles in Europe (Petty, 2000). Koivisto (1963) reported that hens moved from 6 to 24km, but cocks only up to 4km. The main post-breeding dispersal of juveniles appears to occur in late summer/autumn (Petty, 2000).

Two adult cocks radio-tracked in Perthshire in April 1992 remained within 1km of their lek in small home ranges for the rest of the year, and attended the lek in 1993 (Moss & Picozzi, 1994).

Disturbance

Summers *et al.* (2004b) looked at Capercaillie habitat use in relation to tracks in Abernethy Forest, and found that Capercaillies avoided tracks at distances of 61 to 108m. Tree use by Capercaillie was also lower close to tracks with high levels of human use (Summers *et al*., 2004b). A provisional safe working distance of 200-800m from Capercaillie nests and leks is recommended for forestry workers in Britain (Currie & Elliot, 1997).

Capercaillie have also been found to avoid human disturbance (in the form of roads, paths, houses and recently burned areas) in Spain (Suarez-Seoane & Garcia-Roves, 2004). The study compared distances to the nearest source of disturbance between occupied leks and random areas, and occupied and abandoned leks. Occupied leks were found at a distance of 1799m from the nearest road (S.D.= 1053m), compared to 1220m for random points (S.D.= 750m) and 1305m (S.D.= 1012m) for abandoned leks. Similar distances to the nearest path were: occupied leks 987m (S.D.= 593m); random points

709m (S.D.= 681m) and abandoned leks 834m (S.D.= 476m). Distances to the nearest town or building were: occupied leks 1194m (S.D.= 630m); random points 977m (S.D.= 649m) and abandoned leks 911m (S.D.= 338m), and from the nearest area burnt between 1998 and 1999 distances were: occupied leks 2061m (S.D.= 926m); random points 1694m (S.D.= 1054m) and abandoned leks 1538m (S.D.= 944m).

Collision Risk

Collision with fences is a major cause of Capercaillie mortality (Catt *et al.*, 1994, Baines & Summers, 1997, Baines & Andrew, 2003), with collision occurring at a mean rate of 0.9 collisions per km of fence per year (Baines & Andrew, 2003). An annual mortality rate of radio-tagged Capercaillie at Abernethy due to collision with fences of 32% has been recorded (Catt *et al.*, 1994). Grants have been provided to mark or remove fences. Although marking fences reduced Capercaillie collision rates, marked fences still remain an important source of Capercaillie mortality (Baines & Andrew, 2003).

Mapping Capercaillie

Langston and Pullan's (2003) review considered Capercaillie to be at risk of disturbance displacement, collision and direct habitat loss or degradation from wind turbines.

Areas used to identify stratified survey sites for the 2003-04 Capercaillie national survey were plotted (Eaton *et al.*, in press). This was based on a compilation of records from conservation professionals, amateur ornithologists and landowners and managers, and is believed to accurately reflect current distribution (Eaton *et al.*, in press). Just the 'upper strata' were used: that is, sites where Capercaillie were present or breeding in 2003, and/or present in 2000. These were converted into forest blocks over 0.5ha, using the National Inventory of Woodland and Trees (NIWT (Forestry Commission, 2003)). These areas were classified as 'high sensitivity'.

The map of core areas used by the Capercaillie Biodiversity Action Plan Steering Group was not included on the sensitivity map, as it does not identify forest blocks within the outline, so would lead to a lot of unforested habitat being unnecessarily classified as 'high sensitivity'. This is not such a problem for the Capercaillie BAP Steering Group because they are dealing with a single species within a limited range and with very specific habitat requirements within their broadly identified area, whereas the sensitivity map is dealing with multiple species on a larger geographical scale.

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These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in Scotland with respect to a suite of sensitive bird species. Thanks to Rowena Langston who provided helpful comments on an earlier draft of these sensitivity criteria.

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13 Corncrake

Introduction

The Corncrake is a summer visitor to northern Europe from southeast Africa. Britain holds about 13% of the European Corncrake population (Stowe & Hudson, 1992). There have been declines in Corncrake populations over much of Western Europe starting in the nineteenth or early twentieth centuries (Cramp & Simmons, 1980). The UK population of Corncrakes was first noticed to be declining in the late 19th century and had disappeared from much of its former range by 1980 (Green, 2002). The decline has continued since then, from over 2000 singing males in Britain in 1970, to just 750 in 1978, 574 in 1988 and 480 by 1993 (Green, 2002). The British range is now largely restricted to the Northern and Western Isles of Scotland, with 90% of the population being located on the Hebrides (Green & Gibbons, 2000). Until recently, the Corncrake was one of just three species of birds in the UK considered to be threatened with global extinction (Collar *et al.*, 1994, Groombridge, 1994), although it has recently been re-classified as a globally Near Threatened species (BirdLife International, 2004). The main causes of the decline were mechanisation of mowing and intensification of production of hay and silage, and also lack of tall vegetation for cover.

Payments to farmers for Corncrake-friendly management practices, such as Corncrake-friendly mowing and later mowing of silage, began in 1992. The British Corncrake population has increased in recent years, following a sustained decline. Between 1978 and 1993, the population declined at an average rate of 3.4% per year, but between 1993 and 2003, the population increased at an average rate of 5.7% per year, reaching a population of 1040 singing males in core areas by 2004 (O'Brien *et al.*, in prep.). Despite this increase in numbers, the geographical range of the Corncrake has not increased, with Corncrakes being found in 83 10km squares in 1993, 93 in 1998, and 81 in 2003 (Green & Gibbons, 2000, O'Brien *et al.*, in prep.). Whilst there is no direct evidence that the recent population increase is due to the conservation action which was introduced in 1992, modelling of the increase in productivity associated with the Corncrake-friendly management showed that the scale of the conservation action was approximately sufficient to account for the observed population increase (O'Brien *et al.*, in prep.).

The Corncrake is a species of high conservation concern in the UK (Gregory *et al.*, 2002) and is on Annex 1 of the EU Birds Directive. In Great Britain, 204 calling males, or 43% of the British population, occur within Special Protection Areas, with more SPAs proposed (Stroud *et al.*, 2001).

Home Ranges, Site fidelity etc

Mating system

The Corncrake is normally double-brooded; first clutches are laid between mid May until early June, but replacements may continue until August. Corncrakes do not form long-term pair bonds and may be polygynous. Females are solely responsible for incubation and care of the young. Corncrake nests are difficult to locate and so surveys are based on locations of calling males, which sing intermittently during the day and often continuously by night from their arrival at the breeding grounds during April and May until late July or August (Schaffer & Munch, 1993). Radio-telemetry studies have shown that Corncrake nests are usually located within 250m of a male's nighttime calling position (Green & Stowe, 1993).

Dispersal

Estimates of dispersal distance of male Corncrakes in Scotland and Ireland indicated that there was much long distance (>10km) natal dispersal, but that breeding dispersal was relatively low, with only 10% of adult males moving over 10km after their first year (Green, 1999). Most males returned to within a few kilometres of where they were ringed, with most between year movements being less than 1km (Green, 1999).

Two adult males ringed in Ireland in 1992 were trapped the following year within 100m of their original ringing site, and both birds used similar areas for their home ranges as in the previous year (Tyler, 1993). The estimated daily probability of a male apparently leaving the study area on Coll was 0.003 (based on 74 radio tagged males, Green, 2004).

Home ranges

Corncrakes are usually stationary at night but during the day males may wander up to 600m from their singing location, often into areas also used by one or more other Corncrakes (Hudson, Stowe & Aspinall, 1990), returning to more or less the same position each night. Females usually nest and rear their broods close to the singing place of their mates. Radio tracking of nine pairs of Corncrakes revealed that all nests were within 160m (mean= 101m) of the singing place of the male (Tyler & Green, 1996). Second clutches were significantly further away than first clutches (mean= 85m for first clutches compared with 133m for second clutches, Tyler & Green, 1996). Brood rearing areas were also within 200m-300m of the singing place (Stowe & Hudson, 1988, Tyler, 1996).

Most males wandered through the home range of one or more adjacent males, presumably in search of food (Niemann, 1995) or to seek females for extra matings (Skliba & Fuchs, 2004). The home ranges of males in the Western Isles varied considerably, from about 0.04km^2 to over 0.5km^2 (equivalent to circles with radii 0.1km and 0.4km) (Stowe & Hudson, 1991), and these areas are similar to those found by subsequent studies in Ireland. A few males used the same calling sites throughout the season, but others shifted their calling position by up to 0.5km. In South Uist, most of the sites favoured by calling Corncrakes in the first study year were frequented again in the following two years, but usually by different birds (Stowe & Hudson, 1991).

Females radio tracked in the Western Isles wandered widely, but mainly before nest building had started (Stowe & Hudson, 1991). Whilst egg laying, movements of females were more restricted and during incubation they made only short forays from the nest to feed (Stowe & Hudson, 1991). Once the young had hatched, the females moved them away from the nest site, usually further each day. In the Uists, home ranges of females overlapped to some extent depending on stage of breeding, with one female wandering over a range of almost 1km^2, but most ranges were less than 0.3km^2 in size (equivalent to circles with radii 0.6km and 0.3km) (Stowe & Hudson, 1991). These radii are approximations as home ranges will not be circular. One female in South Uist deserted her first clutch and laid a new one about 200m away.

Density

Densities of singing males in Britain and Ireland ranged from 0km^{-2} to 10km^{-2}, the mean being 0.6km^{-2} (Green, 1996). Mean population density varied considerably between regions, with the highest density being 4km^{-2} in north Uist and Benbecula (Green, 1996). Nests have been recorded as close together as 55m but this is probably maximum density (Niemann, 1995).

Disturbance, Collision risk etc

Disturbance of Corncrakes from birdwatchers and photographers has been noted, particularly in the Uists (Williams, Stowe & Newton, 1991), but there has been no work to quantify this.

Collision with overhead wires and lighthouses has been a recognised problem for migrating Corncrakes, which apparently fly at low level, but its scale is difficult to assess (Stowe & Hudson, 1991). Collision risk is presumably lower during the breeding season, although some adults do move between sites occupied at different times during the breeding season. The probability of adult males on Coll leaving the study area is of 0.003 per day, and this equates to a roughly 25% chance of them leaving between May and July. Nothing is known about the heights at which these flights are made (Green, 2004, Green pers. comm.). No mortality from collision was observed in a large sample of radio tagged birds in Scotland and Ireland (Niemann, 1995). In the Uists in the period 1985-1987, one adult was found impaled on a barbwire fence (Stowe & Hudson, 1988).

Mapping Corncrakes

Surveys of the core areas inhabited by Corncrakes (the Hebridean islands and Orkney which held over 90% of the population in the last three national surveys) are conducted annually, and full surveys were conducted in 1993 and 2003. As the British Corncrake population is currently increasing in numbers, if not presently in range, singing males from annual surveys since the 1993 national survey were mapped (1993 to 2003). Another reason for mapping survey data from more than one year is that males may have different singing positions during the course of a breeding season (Tyler & Green, 1996), so it is more likely these alternate calling positions will be included if more than one year of survey data is used. Calling positions were buffered by 250m so that the areas mapped were likely to include nests. These areas were then buffered by 600m, this being at the upper end of the distances documented as being ranged by Corncrakes during the breeding season, and this area was classified as 'high sensitivity'.

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14 Breeding waders: Golden Plover and Dunlin

Golden Plover

Introduction

The global distribution of Golden Plover is restricted to boreal regions of the western Palaearctic, with only a small extension further east (Stroud *et al*., 2001). In Europe, breeding occurs throughout Iceland, Scandinavia and the Baltic States, northern Russia and in northern/upland parts of Britain and Ireland, which form the southern edge of the species' range (Stroud *et al*., 2001). The European population is estimated at between 440 000 and 785 000 breeding pairs (Hagemeijer & Blair, 1997). The core population in northern Europe was relatively stable between 1970 and 1990, but the smaller southern populations have been in decline since the 19th century (Tucker & Heath, 1994). The *apricaria* subspecies is restricted to Britain (22 600 pairs), Ireland (400 pairs), Denmark (6-9 pairs) and Germany (19 pairs) and is in decline (Stroud *et al*., 2004). The recent estimate of the UK population size is 18 300- 33 500 pairs (Thorup, 2006).

In Britain, the species is distributed widely throughout upland areas, with concentrations in northern and western Scotland and the north and south Pennines, and smaller outlying groups breeding in Wales and south west England (Ratcliffe, 1976, Gibbons *et al*., 1993). Two-thirds of the British and Irish breeding population occur in Scotland (Stroud *et al*., 2001). A review of population trends of breeding birds in the UK since 1800 reported that the breeding population of Golden Plover was in decline between 1800 and 1970, and probably also between 1970 and 1995 (Gibbons *et al*., 1996). Numbers in Britain during the 1980s were estimated at 22 600 pairs, and this is the most recent population estimate (Baker *et al*., 2006, based on Reed *et al*., 1985 and Stroud *et al*., 1987). This is compared with 29 400 during 1968-1972 (Sharrock, 1976). Substantial range contractions in Britain and Ireland have been attributed to afforestation, agricultural intensification (Thom, 1986, Boobyer, 1992, Lovegrove *et al*., 1994, Byrkjedal & Thompson, 1998, Fuller & Gough, 1999), reduction of moorland burning and game-keepering (Gibbons *et al*., 1993). These factors have led to extinctions in some areas (Parr, 1992).

The RSPB's Repeat Upland Bird Surveys, which involved the resurveying in 2000 and 2002 of nine study areas (totalling 1 348km^2) first surveyed between 1980 and 1991, found mixed results for Golden Plover, with numbers in one study area having undergone a significant decline, whilst in two others significant increases had occurred (Sim *et al*., 2005). There is, however, evidence of long-term national scale range decrease for Golden Plover, of around 7%, between 1968-72 and 1988-91 (Sharrock, 1976, Gibbons *et al*., 1993). Large range contractions were noted in the south, and particularly south west of Scotland, between the two breeding bird atlases and also by the Repeat Upland Bird Surveys (Sharrock, 1976, Gibbons *et al*., 1993, Sim *et al*., 2005).

Golden Plover breed on heather moorland, blanket bog, acidic grasslands and montane summits. Adjacent pastures are important for feeding adults, and chicks may be moved up to 2km or more to feed in marshy areas rich in invertebrate food (Byrkjedal & Thompson, 1998). Adults are site-faithful and recruits are faithful to their natal area (Parr, 1980).

Breeding densities generally vary from 2-7 pairs/km^2, with exceptional densities of 16 pairs per km^2 having been recorded (Ratcliffe, 1976). Densities in Great Britain are some of the highest within the species' range (Byrkjedal & Thompson, 1998).

The Golden Plover is a species of low conservation concern in the UK (Gregory *et al*., 2002), but is listed on Annex 1 of the EU Birds Directive. Around 26% of the British breeding population of Golden Plover occur within seven SPAs, three of which are in Scotland (Stroud *et al*., 2001).

Disturbance distances

Reported distances at which Golden Plover react to human disturbance range from 50m to 400m (Ratcliffe, 1976, Thompson & Thompson, 1985, Yalden & Yalden, 1989, Yalden & Yalden, 1990, Byrkjedal & Thompson, 1998, Finney *et al*., 2005). Disturbance of Golden Plover has been the subject of a long-term study at Snake Summit in the Peak District National Park. This is an area with an intensively used footpath, the Pennine Way, running through it. Yalden & Yalden (1989) suggested that the high level of recreational disturbance has actually led to population declines in areas of the national park. Finney *et al*. (2005) looked at avoidance of footpaths before and after resurfacing of the footpath in 1994. Prior to the footpath being resurfaced, 30% of people strayed from it, whereas after resurfacing the figure was just 4%. This reduction in people straying from the path coincided with a reduction in avoidance distance of nesting Golden Plover from 200m to 50m (Finney *et al*., 2005). Comparisons were also made with another area of the national park which was subjected to lower visitor pressure. In this area, there was no evidence that Golden Plover avoided nesting close to the path (Finney *et al*., 2004).

Yalden & Yalden (1989) looked at the response of Golden Plover at Snake Summit to an observer approaching and found that adults with chicks alarm-called when approached to within 200m. Disturbance was also found to reduce time spent incubating (Yalden & Yalden, 1989). During the chick-rearing period, adults spent about 11% of their day responding to disturbance, which was estimated to increase energy expenditure by 15%. Less time was also spent brooding chicks (Yalden & Yalden, 1989). Responses to disturbance, such as flushing and reduced time spent brooding, can lead to eggs and chicks being exposed to possible chilling or predation, and impose an energetic cost on adults (Nudds & Bryant, 2000, Bolduc & Guillemette, 2003). However, there was no evidence that chick growth rates and breeding success was reduced close to the Pennine Way footpath (Finney *et al*., 2004).

Disturbance displacement of Golden Plover in response to wind turbines has also been demonstrated, although this has generally been in studies of non-breeding birds. Significant negative effects of wind farms on numbers of non-breeding Golden Plover were found in 21 of 29 studies reviewed by Hötker *et al*. (2004). Disturbance of non-breeding Golden Plover by onshore wind farms has also been demonstrated by a study in the Netherlands (Winkelman, 1992). However, it was not possible to separate effects of the wind farm from other changes that may have occurred, due to the lack of a reference area in this study. The mean minimum distance from wind farms to the nearest Golden Plover found in Hötker *et al.*'s (2004) review was 175m (median= 135m, S.D.= 167, n= 22). Schreiber (2002) observed that Golden Plover avoided wind turbines by 200-300m in winter.

Breeding Golden Plover would thus appear to be affected by human disturbance over distances of a few hundred metres, with a similar magnitude of avoidance distances apparent in response to the presence of wind turbines by non-breeding Golden Plover. However, whether or not this scale of displacement is important biologically will depend upon whether the response is due to the availability of alternative locations and, if so, the energetic costs of moving, together with any implications for survival or future breeding productivity. The duration of the displacement also contributes to whether or not the effect is likely to be detrimental. Signs of habituation have been shown in three of four studies reviewed for Golden Plover, although all of these studies were looking at non-breeding birds (Hötker *et al*., 2004). This indicates that displacement may be a short-term problem, at least for non-breeding birds in relation to wind turbines. Nonetheless, given the present level of knowledge together with potential cumulative effects of multiple wind farms, disturbance displacement is a concern for Golden Plover.

Collision risk

There are few documented cases of Golden Plover colliding fatally with wind turbines. A review of the European literature found two recorded casualties of Golden Plover as a result of collision with wind turbines in Germany since 1989, with a further casualty in both Sweden and the Netherlands (Hötker *et al*., 2004). This relatively low incidence of recorded cases of collision in the (limited) published studies does not necessarily mean that Golden Plover are unaffected by collision with wind turbines. There are aspects of Golden Plover behaviour that indicate they may be at increased risk of collision both when performing breeding display flights, which occur at turbine height (30-100m, Cramp & Simmons, 1983; 15-100m, Byrkjedal & Thompson, 1998), and during foraging flights.

Adults sometimes forage away from the nesting territory and can make foraging flights at night (Parr, 1980, Whittingham *et al*., 2000, Pearce-Higgins & Yalden, 2003), which may increase their susceptibility to collision. Foraging usually occurs away from the moorland nesting habitat during incubation, but on the moorland once broods are present (Whittingham *et al*., 2000). Radio-tracking of European Golden Plovers at night in the UK (Pearce-Higgins & Yalden, 2003) and Germany (Ketzenberg & Exo, 1997), and observational studies in the UK (Gillings *et al*., 2005) have shown that different feeding locations are used at night compared with day-time.

Observations of flight reactions to wind turbines in Schleswig-Holstein indicated that waders reacted at 200-500m from the turbines by increasing flight height or changing direction to fly over or around the turbines (Koop, 1997). Two of three reviewed studies found that wind farms can have a barrier effect for Golden Plover (Hötker *et al*., 2004). Golden Plovers are manoeuvrable but generally fast-flying waders that often occur in large flocks outside the breeding season. It is to be expected that, generally, in conditions of good flight and sight, they will be at low risk of collision. However, in poor flying conditions that hamper manoeuvrability, or when in large flight formations, they are at increased risk of collision and the cumulative effect of large wind clusters or multiple wind clusters is of particular concern.

Dunlin

Introduction

Dunlin have a wide global distribution around the Arctic, and are found in nearly all Arctic regions (Stroud *et al*., 2001). In Europe, they also extend south to temperate regions where they are found in wetland habitats (Stroud *et al*., 2001). The breeding population extends from east Greenland, across the Russian Arctic (Lappo & Tomkovich, 1998) to the Alaskan coast of the Bering Sea. Within Europe, Dunlin breed mainly in the north, this area accounting for less than half of its global breeding range. Its European breeding population is large (over 300 000 pairs) and was probably stable between 1970 and 1990 (although the European wintering population declined markedly). The species was also stable in north west Europe between 1990 and 2000, but declined around the Baltic and Russia, thus the species is given an evaluation of 'Depleted' in *Birds in Europe* (BirdLife International, 2004).

Breeding Dunlin are characteristic of moorland and upland habitats. Concentrations in the UK are found in the Flow Country of Caithness and Sutherland, and on peat moors in the Orkneys, Shetland, Grampians, Pennines and Outer Hebrides (Stroud *et al*., 1987, 1988, Gibbons *et al*., 1993). The machair of the Outer Hebrides is a favoured area, and up to a third of the British population was formerly estimated to breed there (Fuller *et al*., 1986). Dunlin breeding in Britain and Ireland are of the temperate population of *C. a. schinzii,* which also occurs in the Baltic region. The UK breeding population of Dunlin is estimated to be 9 150 – 9 900 pairs (Baker *et al*., 2006, based on Reed, 1985), which represents 83% of the biogeographical population. This is a dated estimate that is overdue revision, but is the best available at present.

The British population is currently considered one of three distinct populations of the *schinzii* subspecies (Stroud *et al*., 2004). There is no recent information on population trends at the national level, although Gibbons *et al.*'s (1996) review of population trends for UK breeding birds suggested that the breeding population of Dunlin was probably in decline between 1800 and 1940, but that there was little change between 1940 and 1995. However, the RSPB's Repeat Upland Bird Survey, which resurveyed nine study areas in 2000 and 2002, that were first surveyed between 1980 and 1991, found widespread population declines of Dunlin (Sim *et al*., 2005). These were large enough to suggest apopulation decline of at least 50% over the last 25 years, leading to the recommendation that the Dunlin's conservation status should be upgraded from 'medium' to 'high' conservation concern (Sim *et al*., 2005).

Several regional studies have also demonstrated declines. In the Flow Country, numbers of Dunlin were estimated to have fallen by 17% due to the afforestation that has occurred there since 1945 (Stroud *et al*., 1987, see also Avery & Haines-Young, 1990), and numbers have continued to decline following the cessation of active afforestation in the area. Whitfield (1997) found that numbers fell on five of 12 sites in Caithness and Sutherland surveyed in the period between 1979 and 1987, and in 1993-4, and the overall numbers fell by 2.4% per year. Afforestation is also estimated to have resulted in the loss of 400 pairs of Dunlin from Wales, the North York Moors and Southern Uplands (Stroud *et al*., 1987). Agricultural intensification has caused additional population losses, for example in Orkney (Booth *et al*, 1984, Ratcliffe, 1990). In the southern Outer Hebrides, the large populations that breed on the machair are in decline, largely due to egg predation by introduced Hedgehogs *Erinaceus europaeus* (Jackson & Green, 2000). The percentage of first nests hatching fell from 59.6% in the period 1985-1987, when Hedgehogs were absent, to 8.8% in the period 1996-1997. Declines have also been recorded in Hedgehog-free areas of North Uist (Fuller & Jackson, 1999).

In contrast to these declines, on peatland sites in Lewis and Harris, where there has not been substantial habitat change, numbers of Dunlin showed a slight increase of 1.4% per year between surveys in 1987 and 1994/5 (Whitfield, 1997, see also Stroud *et al*., 1988).

Breeding densities are often clumped (Jönsson, 1988), with territory sizes of 5.7-6.9ha being recorded in Alaska (Holmes, 1966). Dunlin are site faithful and philopatric (Soikkeli, 1970). Territorial behaviour decreases after pairing and ceases after hatching (Soikkeli, 1970). Broods may travel up to 1km in two days and adults may feed up to 3km from broods (Soikkeli, 1970).

The supspecies of Dunlin that breeds in Britain and Ireland, *schinzii*, is listed on Annex 1 of the EU Birds Directive. The Dunlin is currently a species of medium conservation concern in the UK (Gregory *et al*., 2002), but see earlier text indicating that this may change to "high" at the next revision of Birds of Conservation Concern. Around 74% of the British breeding population of Dunlin occur within eight SPAs, six of which are in Scotland (Stroud *et al*., 2001).

Disturbance distances

Yalden and Yalden (1989) looked at the effect of disturbance on Golden Plover and Dunlin in the Peak District National Park. They suggested that Golden Plover had declined in some areas due to the high levels of recreational disturbance, with Dunlin not having declined in the same areas. They found that breeding Dunlin at Snake Summit, an area with a high level of visitor pressure, reacted to a human observer by alarm calling at a much closer distance than did Golden Plover, not reacting until approached to within about 35m (n= 30, 95% C. I.= 23-38), as oppose to 187m (n= 333). A study in Scotland found similar results, with Dunlin only taking flight when approached to within 20m (Thompson & Thompson, 1985). Distances at which birds take flight in response to disturbance are generally lower than those at which birds alarm call (Yalden & Yalden, 1989). Yalden and Yalden suggested that Dunlin appeared more tolerant to human disturbance due to the fact that they rely on neighbouring Golden Plover as ‘watch dogs’ (as shown by Byrkjedal & Kalas, 1983, Thompson & Thompson, 1985).

Finney *et al*. (2004) looked at Dunlin on Snake Summit before and after resurfacing of the Pennine Way. They found that distribution of Dunlin was affected by habitat and distance from the Pennine Way, with occurrence of Dunlin being more likely closer to the footpath. However, following resurfacing work (which reduced the number of people straying from the footpath from 30% to 4%), the use of areas within 200m of the path increased by about 50%. Median distance between nests and the footpath declined from 175m to 97m (although this was not statistically significant). Thus, it was concluded that Dunlin are affected by disturbance, and that the preference for areas close to the footpath was likely to be due to the habitat in this area (Finney *et al*., 2004).

Collision risk

Males persistently perform display flights during most of the early breeding season, and these are at turbine blade height (10-50m, Holmes, 1966, Cramp & Simmons, 1983), which presents a collision risk factor during the breeding season.

Mapping the breeding waders

Studies at the Dun Law wind farm, in Scotland (Gill 2000a, 2000b) compared the breeding bird populations in the wind farm area plus an 800m buffer with those in a reference area, and found that the waders on the wind farm area showed similar between-year changes in numbers to those on the reference area. Generally, the results indicated that breeding waders were not disturbed by the presence of turbines. However, it is not clear whether turbines might deter settlement by new recruits to the wader breeding population. At Burgar Hill, on Orkney, studies looking at the effects of a three turbine wind cluster found no significant difference in numbers of breeding waders *Charadriiformes* between the year of installation and the subsequent eight years (Meek *et al*., 1993).

In contrast to this, a review of the European literature found that of the species and species groups looked at, only waders and gamebirds displayed reduced numbers during the breeding season in response to wind farms (Hötker *et al*., 2004). Non-breeding waders also generally avoided wind turbines by several hundred metres in the studies reviewed (Hötker *et al*., 2004). Significant negative effects of wind turbines on non-breeding Golden Plover numbers were found in 21 of 29 studies reviewed by Hötker *et al*. (2004). A study investigating breeding densities and distribution of breeding birds before and after installation of four wind farms in Germany, which also compared the wind farm areas with ‘reference areas’, found a negative impact on Redshank within 200m of wind turbines, with there being a similar disturbance effect on feeding birds (Ketzenberg *et al*., 2002). There are records of Golden Plover colliding with wind turbines, and they are likely to be at risk of collision during both foraging flights, which can be made at night (Parr, 1980, Whittingham *et al*., 2000, Pearce-Higgins & Yalden, 2003), and display flights, which occur at turbine blade height (30-100m, Cramp & Simmons, 1983, 15-100m, Byrkjedal & Thompson, 1998). Dunlin also make frequent display flights at turbine blade height (10-50m, Holmes, 1966, Cramp & Simmons, 1983). Barrier effects occurred in two of three studies reviewed for Golden Plover (Hötker *et al*., 2004).

Thus, it is thought that Golden Plover and Dunlin may be sensitive to wind farm development. However, their relatively high population sizes and widespread distribution mean that survey coverage for these species is low (e.g. approximately 3.7% of the Scottish range is covered by the readily available datasets collated for Golden Plover). Golden Plover abundance data were collated from various datasets (RSPB Repeat Upland Bird Surveys, RSPB Grouse Moor Wader Surveys, Moray Moor Waders Surveys, SNH Peatlands SPA surveys, RSPB Upland Grazing Survey, Uists survey, Main Report, Appendix 2). Predictive models were run for Golden Plover in attempt to fill in the gaps in coverage. Generalised Linear Models were run in SAS (SAS, 2003) at the 1km square resolution,

with Golden Plover abundance as the dependent variable. Models were developed, based on Natural Heritage Zone (SNH, 1998), land cover classes from the Land Cover Map 2000 (Fuller *et al*., 2002), an index of grouse moor management (for which the proportion of strip muirburn habitat in a square during Land Cover of Scotland 1988 (MLURI, 1993) was used as a surrogate (after Whitfield *et al*., 2003)), proportion of coniferous forest within 1km and 3km of the 1km square (Forestry Commission, 2003), slope (Ordnance Survey, 2004), elevation (Ordnance Survey, 2004) and latitude and longitude. However, these models did not have sufficient predictive power to justify their inclusion on the sensitivity map (Main Report, Appendix 2).
Another avenue that was investigated for inclusion of waders on the sensitivity map was to include Sites of Special Scientific Interest (SSSIs) that were designated for either Golden Plover or Dunlin. However, information on the SSSI network is not currently available in a suitable format to distinguish SSSIs where Golden Plover and/or Dunlin were present in important numbers as part of a qualifying assemblage of breeding birds, although it is likely to become available in the future.

Although inclusion of data from the New Atlas of Breeding Birds 1988-91 (Gibbons *et al*., 1993) was initially ruled out due to its age, it was decided to include 10km squares with high densities of Golden Plover and Dunlin as ‘medium sensitivity’, indicating areas likely to support higher densities and thus requiring project/site -specific assessment as part of an EIA.

For each of these species, mean tetrad counts from the New Atlas of Breeding Birds 1988-91 (Gibbons *et al*., 1993) per 10km square were calculated. This was done by dividing the sum of all wader counts within tetrads in the 10km square, and dividing this by the number of tetrads surveyed in that10km square (which was a minimum of eight of the possible 25 tetrads). 10km squares with non-zero counts were then divided into five quantiles, and the 20% of 10km squares with the highest mean counts were classified as ‘medium sensitivity’. In the case of Dunlin, taking the top 20% meant arbitrarily excluding equivalent counts in the next quantile so the threshold was set at 22%.

The approach of including 10km squares with high mean counts from the Atlas as ‘medium sensitivity’ was taken, as coverage for these species did not allow for identification of important areas at a finer scale, and it was not thought reasonable to include data of this age as ‘high sensitivity’. Therefore, ‘hotspots’ identified from the Atlas were used to highlight areas where site-specific assessment may identify important areas for these breeding waders. The minimum mean tetrad count per 10km square that was included in this ‘high density’ category for each species is given in Table A1.1.

Table A1.1

	Golden Plover	*Dunlin*
Minimum mean tetrad count	4.6	1

Guidelines for designating Sites of Special Scientific Interest (SSSIs) for these species are that the density must be 1.5 times the regional average density. It is hard to compare these with the mean tetrad counts per 10km square which will yield apparently low mean density estimates, when compared with those per 1km square for species with a patchy distribution.

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15 Arctic Skua

Introduction

The Arctic Skua has a circumpolar distribution, between 82°N and 56°N (Stroud *et al*., 2001). Within this range, it is absent only from the very high Arctic and parts of east Greenland (Cramp & Simmons, 1983, Furness, 1987). The British breeding population is confined to north and west Scotland, and this forms the southernmost part of the Arctic Skua's breeding range. Scottish Arctic Skuas breed on coastal moorland and tundra, occasionally coming up to 10km inland to breed (Cramp & Simmons, 1983). Shetland and Orkney form the Scottish strongholds of Arctic Skua, with 52% of territories being found on Shetland and 34% on Orkney during Seabird 2000 (Mitchell *et al*., 2004). The size of the world population is very poorly known, but is estimated at between 85 000 and 340 000, with 0.6 to 2.5% of this population occurring in Britain (Mitchell *et al*., 2004). The size of the north east Atlantic population is better known and comprises about 15 000 to 30 000 pairs, 6-14% of which nest in Scotland (Lloyd *et al*., 1996, Stroud *et al*., 2001, Mitchell *et al*., 2004).

There have been three complete censuses of breeding seabirds in Britain and Ireland; the first being carried out in 1969-70 (Operation Seafarer), the second in 1985-88 (Seabird Colony Register Census) and the most recent survey in 1998-2002 (Seabird 2000).

There was an apparent increase in numbers of Arctic Skua between 1969-70 and 2000, with the number of Apparently Occupied Territories up by 106% from 1039 to 2136 (Mitchell *et al*., 2004). However, coverage of areas that hold Arctic Skuas was fairly comprehensive during the last two surveys, but less so during Operation Seafarer, which was focused on coastal sites and may have missed Arctic Skua breeding more than 1km inland, making long-term trends difficult to determine (Mitchell *et al*., 2004). An analysis comparing numbers in 1974-75 and 1985-86 in Shetland indicated little change over this period (Ewins *et al*., 1988). Sears *et al*. (1995) again found no change in a more recent census in 1992.

There was a 37% decline in the UK Arctic Skua breeding population between the 1985-87 Seabird Colony Register (SCR) and Seabird 2000 (SB2000), from 3338 to 2136 Apparently Occupied Territories (Mitchell *et al*., 2004). This is likely to be due to declines since the early 1990s; colonies monitored by the UK Seabird Monitoring Programme have indicated that considerable declines have been reported since 1992 (e.g. Walsh *et al*., 1995, Thompson *et al*., 1999). Numbers on Orkney decreased by 31% between 1992 and 1998-2002, and on Shetland by 40% during this period (Mitchell *et al*., 2004). These declines may be linked to poor reproductive output due to low sandeel stock in the late 1980s and early 1990s (Mitchell *et al*., 2004). There is a time lag before low sandeel availability has an effect on the breeding population as age at first breeding for Arctic Skua is 3 to 5 years (Williamson, 1959, Berry & Davis, 1970). Arctic Skua have also been displaced from some breeding areas by the expansion of the Great Skua population (Furness, 1977, Mitchell *et al*., 2004).

An estimated 24% of the British Arctic Skua population, and 3% of the world population, fall within seven Special Protection Areas (Stroud *et al*., 2001).

Breeding System

Arctic Skua are present in Scotland between April and August, and during this period occur in northern waters around Scotland, generally close to nesting colonies (Stroud *et al*., 2001). At the end of the breeding season (August to October), they move south to inshore waters along British and Irish coasts (Stone *et al*., 1995) before migrating to the southern hemisphere. Skuas form a monogamous, generally life long, pair bond, although this is less strong in first-time breeders (Cramp & Simons, 1983). Breeding pairs are either solitary or loosely aggregated in varying degrees such that some authors refer to colonies. There were aggregations of 50 pairs on Fair Isle (Williamson, 1949), and of 200-300 pairs and 54 pairs on Hermaness (Shetland) in the 1920s and in 1974, respectively (Pitt, 1923, Glegg, 1927, Albon *et al*., 1977).

Where breeding is solitary, most foraging occurs within the territory. It is the core area that is defended most vigorously throughout the season (Maher, 1974). In colonies, smaller territories are defended and used for heterosexual behaviour and care of young, but foraging is performed elsewhere (B L Furness, cited in Cramp & Simmons, 1983). On Shetland, each territory has a look-out mound, these are an average of 77m apart, and never less than 30m, with the nest up to 20m from the mound (Perry, 1949). Territory sizes of about 1.29km^2, with a core area 400-600m in diameter have been recorded in Alaska (Maher, 1974), and of about 20-100m on Shetland (Furness, 1977). Pairs use the same territory each year and usually build the nest on, or close to, the previous site (Dementiev & Gladkov, 1951, Maher, 1974). Replacement clutches are laid in the original territory, but not usually near the first nest (Furness, 1977). A clutch of two is laid, Arctic Skua are single-brooded, laying replacements following failure at the egg stage (Cramp & Simmons, 1983). Incubation takes 25-28 days (Cramp & Simmons, 1983).

Most Arctic Skua breeding sites have been established for many decades or even centuries (Mitchell *et al*., 2004). Few new colonies have formed and the breeding range has remained remarkably static (Mitchell *et al*., 2004). Arctic Skua usually return to exactly the same territory each year to breed (Mitchell *et al*., 2004).

Nearest-neighbour distances, densities etc

Distances between nests of solitary pairs can be several kilometres (Cramp & Simmons, 1983), and between aggregated/colonial nests distances of 30-90m (Perdeck, 1963) and of 200-400m (Farrar, 1950) have been recorded on Shetland.

Fair Isle, Foula and Papa Stour are the most densely populated areas in Shetland, all holding just over 12 Apparently Occupied Territories (AOTs) per km^2, with the lowest densities being in the central, west and northern mainland of Shetland (Sears *et al*., 1995). The other eight of the eleven sites looked at had densities of less than 4 AOTs per km^2, and the mean density across the eleven was 1.3 AOTs per km^2 (Sears *et al*., 1995). Nest densities of 100-211 per km^2 were recorded on Foula (Shetland) in 1975 (Furness, 1977).

Disturbance

At Burgar Hill, on Orkney, Scotland, studies of distance effects in relation to the 3-turbine wind cluster found no significant difference in numbers of breeding pairs of Arctic Skua between the year of installation and the subsequent 8 years (Meek *et al*., 1993). However, in view of the Arctic Skua's longevity and site faithfulness, it is possible that there may be a considerable time lag before any effect on nest location might become apparent, if territories are still being held by the original birds. It could be that following vacation by these birds, new territory holders do not move in to replace them.

Collision Risk

The Environmental Impact Assessment for the offshore wind farm at Horns Rev, Denmark, states that 'for gannets, terns and skuas, it is not possible to completely rule out the potential risk of collisions by these birds when foraging in the area' (Noer *et al*., 2000). Skuas obtain a substantial part of their food by stealing from gulls and terns. This behaviour often involves vigorous and intense flights where skuas chase their victim until they regurgitate their stomach contents. Such chases usually occur 5-20m above sea level but may go far up in the air. A potential risk of collision for both skuas and their victims exists if such chases take place within the turbine area.

An overhead wire on Papa Westray reserve resulted in several Arctic Skua mortalities through collision (Meek, pers. comm.). Onshore, Arctic Skuas could potentially be at risk of collision with wind farms during both foraging and display flights.

Mapping Arctic Skuas

The Arctic Skua has been included on the sensitivity map, despite not being an Annex 1 species, as it is a colonial breeder with relatively low SPA coverage (24%). There may therefore be important Arctic Skua colonies that are unprotected by the designated site network.

Arctic Skua may be sensitive to collision risk during foraging and display flights; most foraging flights and heterosexual behaviour for solitary nesters occurs within the territory, heterosexual behaviour for colonial nesters occurs within the territory, but foraging flights occur elsewhere. As there is little information on the length of these foraging flights, and they are unlikely to be equal in all directions from the nest, and differ between coastal nesters and birds further inland, it was decided not to attempt to account for these on the map by use of buffers. Areas of high foraging activity should be picked up on during Environmental Impact Assessments.

There were 2136 AOTs during Seabird 2000, thus a ‘numerical cut-off’ point of 21 AOTs could be considered suitable for allocating a high sensitivity rating, as 1% of the population. However, the problem with this is that there have been major declines in Arctic Skua numbers even since Seabird 2000, making 1% of the population difficult to estimate. Thus, a precautionary approach was taken, and any 1km squares with 10 or more Arctic Skua in were classified as ‘high sensitivity’. As territory sizes are relatively small (with a diameter of about 20-100m on Shetland, Furness, 1977) and static (Mitchell *et al*., 2004), no buffers were applied to these squares.

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16 Nightjar

Introduction

The British Nightjar population has been declining in numbers and range throughout most of the 20th century. The Nightjar's British range has contracted from the north and west, with the population now being concentrated in areas of lowland heath along the south coast and increasing in recently restocked areas within coniferous plantations in the south and east. Large declines have also occurred in most other European countries.

National surveys are conducted regularly, based on numbers of churring males due to difficulties with finding nests. The 1981 survey counted 1784 churring males and estimated that about 2100 pairs were breeding in Britain (Gribble, 1983). Although data from the 1957-8 survey for Scotland was sparse, the Western Isles, Caithness and Moray had all lost their Nightjars by 1981, with the Scottish population being concentrated in Dumfries and Galloway (Gribble, 1983). The UK decline up to this point is likely to be largely due to reduction in area of and fragmentation of lowland heath habitat, formerly from afforestation for conifer plantations and conversion to arable and then from scrub invasion due to lack of grazing and better control over heath fires, and building development.

The 1992 national survey located 3039 churring males, and this gave a British population estimate of around 3400 males (Morris *et al.*, 1994). This was an increase of around 75% on the estimate from the 1981 national survey. There was no corresponding increase in range (Morris *et al.*, 1994). The Scottish range also stayed fairly constant between the two surveys. The main explanation given for this partial population recovery was changes in forest structure. The expansion of commercial forestry after 1945 led to habitat loss for the Nightjar once the forests matured, as forests become too dense to provide nest sites after 10 to 20 years. Harvesting of these forests during the 1980s meant that Nightjars were able to nest in the clearfells and restocks. Additional woodland clearings were created by the gales of 1987 and 1990. The fact that the proportion of records in conifer plantations increased between 1981 and 1992 supports this theory (Morris *et al.*, 1994).

The 2004 national survey counted 4131 males, giving a population estimate of 4606 (Conway *et al.*, in press). This was an increase of 36% since the 1992 national survey, and there was also a range increase of 2.6% in terms of occupied 10km squares. However, the increase was not uniform across the range, and there was evidence of population declines and range contractions in north west Britain, including north Wales, north west England and Scotland. Fifty seven percent of the Nightjars surveyed in 2004 were associated with forest plantations and 59% with heathland. The continued overall increase was probably down to habitat protection, management and restoration of heathland, and forestry providing continuity of available clearfell or young plantations in conifer forests. Scotland is the only area where the range has decreased between the two latest surveys, from 18 10km squares occupied in 1992 to just seven in 2004. There was also an estimated population decline of 23% (Conway *et al.*, in press). All males recorded in Scotland were located within forest plantations and fewer birds were located on clear-felled areas than in 1992. It is possible that declines in Scotland are due to changes in breeding habitat quality, but other factors, such as the availability of wetland edge habitats and the general availability of foraging habitat could be equally limiting (Rollie, pers. comm., cited in Conway *et al.*, in prep.).

The Nightjar is a species of high conservation concern in the UK (Gregory *et* al., 2002) and is protected under Annex 1 of the EU Birds Directive. Fifty three percent of the British Nightjar population (1 785 churring males) occur within the ten Special Protection Areas designated for the species in the UK, all of these are in England (Stroud *et al.*, 2001).

Territories, Site Fidelity etc

Breeding system

In East Anglia, the first males usually arrive in the second week of May and start churring (Berry, 1979). Males roost about 50m (occasionally 100m) from the incubating female and show remarkable fidelity to roost sites (Berry, 1979). In the evening, on leaving the roost area, a male Nightjar will chur at various regular song posts in his territory and then either relieve the female at the nest or feed (Berry, 1979).

Birds are faithful to a nest site area (though not the precise nest site) year to year (Berry, 1979). Second clutches are generally thought to be relatively uncommon (e.g. 24% of territories at Minsmere in Suffolk, Berry, 1981), and particularly so further north in the range. They are usually laid relatively close to the first clutch, with the mean distance being 124m (range 70-160m) in Dorset (Cresswell, 1996). Males at Minsmere arrived between 10th May and 1st June and females between the 11th May and 16th June (Berry, 1981). First eggs were laid between the 17th May and 28th June (Berry, 1981). Young hatch after 17 to 21 days incubation and are probably dependent on their parents for about 30 days (Berry, 1981).

Densities

Densities are generally higher in young plantations than on heathland sites, indicating possible preference for this habitat. There may be a long-term decline in densities on heathland; Lack (1932) found there was a mean density of 1 pair per 12 hectares at Kelling Heath in Norfolk. The average density at Minsmere from 1948 to 1977 was 1 per 12.8ha, but by 1977 it was 1 per 19.6ha. In Dorset, a comparison was made of ten forestry sites (up to 20 years old) and ten heathland sites (referred to in Morris *et al*., 1994). The mean density of singing males on the forestry sites was 1 per 7.7ha compared to 1 per 17.3ha on heathland. The 2004 Dorset Nightjar survey found an overall median density of churring males of 1 per 20ha (Lake, 2004). However, densities were higher on heathland (median= 1 per 25ha) than forestry (median= 1 per 16.7ha). This difference was significant, and remained so when non-suitable forestry habitat was excluded from the analysis, when the median densities were 1 per 20ha for heathland, and 1 per 16.7ha for forestry sites. The greatest densities by far were found at heathland:forestry interfaces, and densities in forestry were highest in 1-3 year old plantation (Lake, 2004). Ravenscroft (1989) looked at restocks between 1 and 5 years on the Suffolk coast and found densities for churring males ranged between 1 per 8ha and 1 per 17ha, but were usually between 1 per 11 and 14ha. Cadbury (1981) found densities on Suffolk heathland of 1 per 21.3ha. Berry (1979) found densities on heathland in East Anglia of 5.9-25.2ha per pair. On Sandlings heathland, an average density for one site over five years was 21ha per bird (range 17-26ha, Ravenscroft, 1987). Brünner (1978) found that in Bavarian forest birds occupied 11ha each. Invasion by dense scrub is probably the primary cause of reduced heathland densities. Disturbance may be important too (Liley & Clarke, 2003).

Ravenscroft concluded there was a minimum area of clearing required to support Nightjars (about 10ha, equivalent to a circle of radius 178m). Wichmann (2004) found that territories generally centred on a large clearing with an area of at least 0.7ha, and over 50m wide, in a population of Nightjars in an Austrian pine forest. Minimum areas of clearing required cited in other studies vary widely from 1.5 to several hectares (Abs, 1963, Brünner, 1978, Cramp, 1988, Ravenscroft, 1989).

Territory sizes

Territory sizes are smaller than densities would suggest, with each males' singing post in East Anglia encompassing an area of 5 to 6 ha (equivalent to circles of radii 126m and 138m, Berry, 1979). This is because territories are not evenly spaced. Sizes of breeding territories on heathland vary considerably; different studies recording 7-30ha, 21ha and 12-20ha (cited in Morris *et al*., 1994). Average territory size at Minsmere in 1976 and 1977 was 5.3ha (Berry, 1979). Mean nearest-neighbour distances were 164m at Dersigh and 172m at Minsmere (Berry, 1979). Of eight territories surveyed by Cadbury (1981), five were relatively small, ranging from 1.5 to 6.1ha, and in discrete areas with clumped distribution of registrations indicating favoured song posts. The remaining three territories were larger as they included outlying song posts. These were evenly spaced with mean nearest-neighbour distances (between territory centres) of 487 +/- 116m (range 280-600m, n= 7).

After the main song period at dusk, birds were observed moving out over open heath, presumably to feed. Berry (1979) states that Nightjars do not appear to defend their feeding range and communal feeding is frequent. Wichmann (2004) found that in a population of Nightjars living in an Austrian pine forest, a narrow strip of forest around the territories was defended by the male. The song perches ,and with them the boundaries of territories, were sited in tree plantations at the edge of the clearings and were at most 30m from them. Territorial disputes occurred only when two clearings were close to each other.

Collision Risk

Nightjars would be mainly at risk of collision during foraging or display flights. Nightjars are crepuscular and so would be expected to have generally good night vision. There is wide variation in the lengths of foraging flights found by different studies. Alexander and Cresswell (1990) radio-tracked Nightjars in Dorset and found that mean distance to foraging area was 3.1km +/- 1.2km. The maximum distance recorded was 5.8km, but most flights were between 2 and 4km from the nesting area. This agrees with what was found by Glutz von Blotzheim (1962) and Schlegel (1967) who described Nightjars foraging several kilometres from suitable nesting areas prior to the laying period. The most favoured foraging habitats during the breeding period were deciduous or mixed woodlands and gardens/orchards (Alexander & Cresswell, 1990). Despite their greater availability, conifer plantations and improved grassland/arable were little used, and there were no records of visits to dry heath.

This is different to what was found by a radio-tracking study of breeding Nightjar in Norfolk and Suffolk (Bowden & Green, 1994) where Nightjars did not make such long excursions. Nightjars in this study did most of their foraging within 1km of the centre of the breeding territory, and did not forage further than 2km from the centre (Bowden & Green, 1994), in comparison with the 6km trips of the Wareham forest birds.

Wichmann (2004) also found that Nightjars in his study area in an Austrian pine forest hunted largely inside the pine forest and within their territories. Despite intensive search with night vision equipment and binoculars, only a single female was observed hunting outside the territories, whereas a number of chance observations were made within territories (Wichmann, 2004).
It was not possible to say how much Nightjars may have encroached on each other's territories to hunt. In sites where prey is abundant, Nightjars have been reported to hunt together in groups (Lack, 1932, Berry, 1979). Nightjars often use the same feeding sites for many nights in succession (Creswell, 1996).

The variation in length of foraging flights is likely to be related to habitat surrounding the breeding territory and to local food abundance. Hunting strategy could also change during the course of the breeding season (Schlegel, 1967, Alexander & Cresswell, 1990).

Flights are generally low over, for example, bracken or heather, but along the forest edge, birds fly close to the top of the canopy (Cramp *et al*., 1985). It is not clear whether Nightjars fly at sufficient height to be at risk of collision with turbines (mean lowest sweep of rotor= 27m, n= 84, calculated from SNH data of wind farms in Scotland and this lowest sweep height is likely to be getting higher as new, larger turbines are developed). There is little published evidence of flights above this height, although one case of an unpaired Nightjar making a patrolling flight at 10-15m above trees has been documented (Cramp *et al.*, 1985). However, Nightjars approach their prey not horizontally, but from below in steep flight (Schlegel, 1967, Lehtonen, 1969, Bühler, 1987). Palmer (pers. comm.) has observed high spiralling flights in Nightjars and Rollie (pers. comm.) observed a flight to a height over 30m; presumably these birds were chasing prey. Wotton (pers. comm.) has frequently seen birds flying above mature conifer height (over 20m+) during display flights, flying from one churring spot to another, and on foraging flights. Nightjars are however quite manoeuvrable in flight (Cresswell, 1996), and also have enhanced night vision, both of which are likely to reduce their risk of collision.

Disturbance

Nightjars are sensitive to disturbance at breeding sites (Murison, 2002, Liley & Clarke, 2003, Woodfield & Langston, 2004). Liley and Clarke (2003) looked at 36 heathland sites in Dorset and found that Nightjar densities had a strong negative relationship with surrogates for disturbance (amount of developed land within 250m, 500m and 750m of the heath, and number of houses surrounding the heath). Murison (2002) found that urban heaths supported fewer territories and that path proximity and the amount of footpath surrounding nests were both significant predictors of Nightjar breeding success. Path effects correlated strongly with nest failure up to 225m from the path edge, but significant effects of paths were found up to 500m, which was the outer range investigated (Murison 2002). Woodfield and Langston (2004) looked at four Dorset heaths with particularly high levels of access in order to look for a mechanism for these relationships. There were non-significant trends for Nightjar nests to be in areas of fewer footpaths, further from main access points, and surrounded by higher vegetation cover. There were relationships between hatching success and proximity to footpath. These were not statistically significant, but this may have been a result of lack of power.

Mapping Nightjars

1km squares in Scotland in which churring males were found during the 2004 national survey were mapped.

Ravenscroft (1989) calculated the minimum clearing needed to support Nightjars as being a circle of about 200m radius. Murison (2002) found that path effects correlated strongly with nest failure up to 225m from the path edge. Murison (2002) also noted greater total path length within 500m of the nest being associated with significantly higher nest predation, with there being a similar effect of housing within 500m of heathland (Liley & Clarke, 2003). However, it is not possible to identify a 'cut-off' distance at which disturbance is important from these last two studies.

There is little published information on flight height in Nightjars, however there is a large body of anecdotal evidence of Nightjars making flights (presumably foraging flights) at a sufficient height to be at risk of collision with wind turbines.

There is fairly wide variation in distances of foraging flights reported in different study areas, with Alexander and Cresswell (1996) finding a mean foraging flight distance of 3km in Dorset, and Glutz von Blotzheim (1962) and Schlegel (1967) also describing Nightjars foraging several kilometres from suitable nesting areas prior to the laying period. Bowden and Green (1994) however, found that most foraging was conducted within 1km of the territory centre and that all foraging flights were less than 2km in Norfolk and Suffolk, suggesting foraging distances may be lower elsewhere in the UK. Wichmann (2004) found that Nightjars in his study area in an Austrian pine forest hunted largely inside the pine forest and within their territories. Foraging distances will be dependent on the proximity of suitable feeding habitat to the nest, so it is not unduly surprising that there is variation between these studies.

Based on this limited information on disturbance distances and collision risk, a buffer of 1km around 1km squares containing Nightjar will be classified as 'high sensitivity' (which should encompass most of the lower estimates of foraging flight distances found by Bowden and Green (1994)). A further buffer of 2.5km around the square classified as 'medium sensitivity' (given the larger mean foraging flight distance of 3km found by Alexander and Cresswell, but taking into account the fact that flight distances elsewhere in the UK may be shorter). Foraging flights extending further than these buffers may occur, these would need to be looked for in site-based assessments.

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17 Chough

Introduction

The Red-billed Chough *Pyrrhocorax pyrrhocorax* has a highly localised and fragmented distribution in the Palaearctic region, and is declining in numbers over much of its European range (Guillou, 1981, Bignal & Curtis, 1989, Blanco *et al*., 1991). The European breeding population is estimated at less than 110 000 pairs (BirdLife International, 2004), with over three quarters of these being concentrated in Spain, Greece and Italy (Tucker *et al.*, 1994). There is little evidence of interchange between breeding populations, and previous contractions and declines have eventually resulted in extinction, e.g. from England in the 1970s and Austria in the 1950s (Stillman *et al*., 1998). The European breeding population underwent a large decline between 1970 and 1990, and although populations were stable across much of the Chough's European range between 1990 and 2000, key populations in Spain and Turkey declined, and the population underwent a moderate decrease overall (BirdLife International, 2004). These declines are almost entirely attributed to the loss of traditional low intensity livestock farming (Bignal & Curtis, 1988, Tucker & Heath, 1994). Stroud *et al*. (2001) estimated a biogeographic population size of 12 265.

In the UK and Ireland, the Chough is largely restricted to areas of low-input agricultural land. The Chough's British range has contracted over the last 300 years and the breeding population is now restricted to western coasts and islands (Gibbons *et al*., 1993). The population has also been decreasing in numbers over this time (Warnes, 1983). The reasons for the population decline are not clear but may be related to persecution, changes in climate and the intensification of farming practises (Bullock *et al.*, 1983, Warnes, 1983). The numbers of Chough in Scotland represent about 24% of those in the UK and Isle of Man, and 6% of those in Britain and Ireland, and have tended to fluctuate over the last 20 years. Numbers increased between 1982 and 1986, but then decreased from 1986 to 1992 (Cook *et al*., 2001). Within the UK, persecution has had an impact on Chough for much of the early part of the 20th century and the population decline has continued in recent decades because of agricultural intensification (Bignal *et al*., 1997).

The Inner Hebridean islands of Islay and Colonsay have been the bird's major stronghold in Scotland for many years (Thom, 1986, Monaghan *et al*., 1989). Between 1986 and 1998, over 90% of the population was found on these islands. The Chough's Scottish breeding population increased form 75 pairs to 105 pairs between 1982 and 1985, although some of this increase may be attributable to better survey coverage. This increase was followed by a decline to 88 pairs in 1992 and just 66 pairs by 1998 (Cook *et al*., 2001).

The last comprehensive national survey of the Scottish Chough population was in 2002 and this estimated the total population size at 256 birds, with 83 pairs being found at nest sites (Finney & Jardine, 2003). This was an increase of 26% since the previous national survey in 1998. Despite this increase, the Scottish range of the Chough has declined. The loss of pairs on Mull and Jura mean that the population is now restricted to the islands of Islay, Colonsay and Oronsay, with the exception of a single breeding pair in Galloway (Finney & Jardine, 2003).

The Chough is a species of medium conservation concern in the UK (Gregory *et al*., 2002) and is listed on Annex 1 of the EU Birds Directive. The majority of birds are concentrated within a very small area, making the population vulnerable to threats such as changes in land use. Around 33% of the British breeding population of Chough are within nine SPAs (Stroud *et al*., 2001). Of these, eight are also designated for non-breeding Chough.

Ecology and Breeding System

Most sub-adult Choughs spend their first two years as part of a flock or flocks with other sub-adults. These flocks forage widely but not randomly, always focusing on some key areas of sand dune, machair, sand grassland or limestone grassland, and form communal roosts at night, typically on rock faces (Still *et al*., 1987). Most Choughs first breed successfully in their third year of life (Bignal *et al*., 1997). Established breeders maintain contact with their nest sites and their home range foraging area throughout the year, roosting at or close to their nest. Early in the year, Chough sometimes congregate to forage in small flocks, sharing their home ranges without territorial disputes.

Nest building begins in March and egg-laying in late March or early April. The Chough uses a variety of nest sites, including ledges in caves, crevices and both used and abandoned buildings (Finney & Jardine, 2003). Most nests are 10-30m (range 1-250m) above the ground or sea (Cramp & Perrins, 1994). Each pair may have several potential nest sites, and in any year nests may be rebuilt at more than one of these before they decide which one to use (Bignal et al., 1997). With the onset of egg-laying, pairs become strongly territorial and will not tolerate adjacent pairs in their home ranges (Bignal et al., 1997). A mean clutch size of 4.6 was found on Islay (S.D.= 1.1, n= 63), the most common clutch size in Scotland is five (Cramp & Perrins, 1994). While the female incubates, the male defends the home range, forages within it and feeds the female at or close to the nest. Both parents make regular foraging trips within their home range. Young leave the nest after 6 weeks. Second broods are extremely rare (Cramp & Perrins, 1994).

Dispersal and site fidelity

Re-sightings of colour-ringed birds have shown that male Choughs tend to nest close to their natal site whilst female Choughs disperse to breed, with natal dispersal distances generally being less than 4km for males and more than 10km for females (Bignal *et al*., 1989). Post-fledging dispersal distances of over 50km have been recorded (Wernham *et al*., 2002).

Choughs are very site faithful and once established at a nest site will continue to attempt to breed there despite repeated failures, for instance due to disturbance, predation or management changes (Scottish Chough Study Group, 1995).

Disturbance

A study of radio-tagged Alpine Choughs *Pyrrhocorax graculus* in Italy found that human developments were avoided by foraging Chough, and that use of a meadow was reduced in 1990 following extensive ski development (Rolando & Pattersen, 1993).

However, Cramp and Perrins (1994) state that the Red-billed Chough is often less shy than other corvids in Britain, sometimes allowing approach to within 10m. Detailed studies of the reaction of Choughs to tourists in South Wales (Owen, 1989) found a mean minimum flushing distance of 35m (S.D.= 6, n= 101), whereas minimum distances as low as 9.5-11m were found elsewhere in Britain (Cramp & Perrins, 1994). Chough have also been found to be tolerant of disturbance at active quarries in North Wales (Johnstone, pers. comm.).

Densities, territory sizes, foraging ranges

Robertson *et al*. (1995) found a mean density of 1.3 Chough km^{-2} in County Donegal in Ireland; this is comparable to the Irish national average (Berrow *et al*., 1993). In Britain, nests are usually several hundred metres or more apart where densities are relatively high, such as on the Isle of Man, whilst in Wales, nests were on average 1.4km (n= 32) apart in good habitat, but 2.6km (n= 27) apart in poorer habitat (Bullock *et al*., 1983). The highest densities in Britain are probably on Islay with up to 0.33 nesting pairs per km^2 (Monaghan *et al*., 1989), with densities being much lower in a mountainous area of North Wales (Holyoak, 1972).

Three quarters of 1km squares looked at in the 2002 national survey with possible/probable/confirmed breeding records contained only one site, indicating that Choughs mainly nest at low densities in the UK and Isle of Man (Johnstone *et al*., in press). Densities were significantly higher in the Isle of Man than in Wales and Scotland, and in a few places on the Isle of Man and in Wales nest sites were clustered on favoured sea cliffs and quarries within 1km squares (Johnstone *et al*., in press). This 'colonial' breeding has been reported previously for Chough on the Isle of Man (Bullock *et al*., 1983) and occurs more markedly elsewhere in Europe (e.g. Blanco *et al*., 1998). However, the UK and Isle of Man Chough population still consists mainly of solitary breeders nesting at low densities with exclusive home ranges (Johnstone *et al*., in press).

Territory around the nest is defended, with pairs being observed driving conspecifics away when they intruded within several hundred metres of the nest sites (Holyoak, 1972).

The radii of home ranges in County Donegal ranged from 1.2km to 2km (n= 5, Robertson *et al*., 1995). The five breeding pairs observed showed a preference for foraging in machair and maritime turf, with rough grass also being important, and pairs avoiding heather and improved pasture. These foraging preferences were similar to those from previous work on habitat selection by Choughs in Ireland (Bullock *et al*., 1983).

Cramp and Perrins (1994) state that 'much but by no means all feeding is carried out within the home range away from the nesting territory'. A study of three pairs in Islay found that they had a maximum foraging flight distance of 1.2km, 0.8km and 0.6km respectively (Bignal *et al*., 1996). A more recent study in Islay, conducted during the 1998 national survey, found Choughs foraging a mean distance of 595m (+/-71 S.E., n= 38) from the nest (Cook *et al*., 1999). Inland in North Wales, several pairs were regularly seen flying from nests to collect food 2-3km away (Holyoak, 1972), although in Welsh coastal areas distances between nests and observed feeding places averaged 0.7km (n= 58, Bullock *et al*., 1983). Whitehead *et al*. (2006) looked at 14 pairs of breeding Chough in North Wales and found that the density of habitat use decreased asymptotically with distance from the nest, such that 95% of foraging flights occurred within 1km of the nest. A study in south west Ireland looking at birds from 12 sites in 2004 found that birds spent 95% of their time within 1.3km of the nest (Gray *et al*., 2004).

Breeding birds often flock with conspecifics, breeders and non-breeders, flying high into the air and up to several kilometres from the nest site (Holyoak, 1972). Members of flocks often soar for short periods and frequently perform skilful acrobatics, commonly making earthward swoops from high up (Cramp & Perrins, 1994).

Mapping Chough

Although Chough have not been shown to be at risk of collision, and their sensitivity to disturbance does not seem particularly high, the extremely small and localised nature of the Scottish breeding population justifies a precautionary approach.

Studies of breeding Chough in North Wales have found that most of their foraging flights (95%) are within 1km of the nest (Whitehead et al., 2006). Other studies have found mean foraging distances of 0.6km (Cook et al., 1999), and 0.7km (Bullock et al., 1983), or maximum distances for individual pairs of 1.2km, 0.8km and 0.6km (Bignal et al., 1996). A study in south west Ireland found that birds spent 95% of their time foraging within 1.3km of the nest (Gray et al., 2004). SNH guidelines for defining SPA boundaries use a distance of 1km from the nest site, excluding only unsuitable habitat within this radius and including areas beyond it known to be used by foraging chough at the site in question.

Based on this, nest sites from the 2002 national survey (which are at the 1km square level of resolution) were buffered by 1km, and this area classified as 'high sensitivity'.

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Appendix 2. Golden Plover modelling

Models were developed with the aim of producing predicted abundances of Golden Plovers throughout their Scottish range that could be used in the development of a sensitivity map to provide locational guidance for wind farms with respect to a suite of sensitive bird species (see main report). Generalised Linear Models (GLMs) were developed in SAS (SAS, 2003) to predict Golden Plover abundance at the 1km square level of resolution. Minimum adequate models were found by a stepwise deletion process from the maximal model.

Golden Plover abundance data from 1 890 1km squares were collated (Table A2.1). These were used to predict abundance throughout the Golden Plover's Scottish range. This was taken as 10km squares where Golden Plover were present in the New Atlas of Breeding Birds 1988-91 (Gibbons *et al*., 1993), plus any extra 10km squares with possible, probable or confirmed Golden Plover breeding records since 1988 downloaded from Merlin (the RSPB data library). The resulting range consisted of 51 334 1km squares (following removal of 1km squares that fell in the sea). Survey coverage was 3.7%. SNH's 'Natural Heritage Zones' (NHZs), 21 biogeographic zones of Scotland (Figure A2.1, SNH, 1998), were used for the parameter representing region. However, surveyed squares were very clumped in distribution, such that nearly all of the data were from just four of the 21 Natural Heritage Zones (NHZ 3, Western Isles; NHZ 5, The Peatlands of Caithness and Sutherland; NHZ 11, Cairngorm Massif and NHZ 19, Western Southern Uplands and Inner Solway). All other NHZs contained just 93 surveyed 1km squares in total, and fewer than 50 in any one of them. Therefore, models were run both with and without the term 'Natural Heritage Zone'. Models including the NHZ term were created using counts just from the four NHZs mentioned above, and were for prediction within these zones. Those without the NHZ term were for abundance predictions in the other 17 natural heritage zones. Where included, Natural Heritage Zone was included as a factor.

Other terms incorporated in the model were:

- Percentage of a number of **land use categories** from Land Cover Map 2000 (see Table A2.2, Fuller *et al*., 2002). Some of these were lumped into ecologically similar habitat categories, and/or according to knowledge of Golden Plover habitat associations, to reduce the number of potential explanatory variables and minimise the risk of colinearity between variables;
- An index of **grouse moor management** (for which the proportion of 'strip muirburn' habitat recorded in a square during Land Cover of Scotland 1988 (MLURI, 1993) was used as a surrogate (after Whitfield *et al*., 2003));
- **Proportion of coniferous forest within a 1km buffer zone** around the square (calculated using the National Inventory of Woodland and Trees (Forestry Commission, 2003));
- **Proportion of coniferous forest within a 3km buffer zone** around the square (as above);
- Proportion of the 1km square with a **slope** of less than 5° (calculated using OS Land-Form PANORAMA® data (Ordnance Survey, 2004));
- **Mean elevation** (calculated using OS Land-Form PANORAMA® data (Ordnance Survey, 2004));
- **x** and **y** co-ordinates and
- All two way interaction terms with Natural Heritage Zone, plus x*y and quadratic terms for x, y and elevation.

In the models including Natural Heritage Zone, if over 95% of the values of any land cover category was 0, then all of them were set to 0 in order to increase the power of the model.

1 Model including Natural Heritage Zone

See Table A2.3 for results of the model including Natural Heritage Zone.

Twenty percent of the Golden Plover counts were excluded from the dataset used to build the model, so that they could be used as a validation dataset. When used to predict abundances for this validation dataset, the model had an R^2 value of 0.22.

To determine whether this was accurate enough to predict Golden Plover 'hot-spots' for the sensitivity map, the ability of the model to predict whether or not a 1km square contained two or more Golden Plover was tested. This cut-off point of two was chosen following a review of the 'critical density' of Golden Plover for SSSI designation, which is equal to 1.5 x the regional mean density (in pairs per km^2). Values for five regions of Scotland are given in Table A2.4. The mean critical density for the five regions was 1.9 pairs per km^2, hence the choice of a cut-off point of two to test the models. For the validation dataset, counts of less than two were predicted correctly 92% of the time. However, counts of two or more Golden Plover were predicted correctly just 43% of the time. This was not considered sufficiently accurate to allow for inclusion of predicted values on the sensitivity map.

2 Model excluding Natural Heritage Zone

See Table A2.5 for results of the model excluding Natural Heritage Zone.

Again, a validation dataset made up of twenty percent of the Golden Plover counts was excluded from the dataset used to build the model, this yielded an R^2 value of 0.14. This validation set was selected randomly from all counts, and thus is likely to comprise mainly squares within the four NHZs where most surveys occurred. Thus, this figure is likely to be an overestimate of the model's predictive power across the rest of the Golden Plover's Scottish range (as region has an influence on abundance).

As before, predictions for the validation dataset were inspected to test whether the model could predict Golden Plover 'hotspots'. Counts of less than two were predicted correctly 93% of the time. However, counts of two or more Golden Plover were predicted correctly just 31% of the time. This was not considered sufficiently accurate to allow for inclusion of predicted values on the sensitivity map.

A model averaging approach was also investigated. Given the relatively large number of variables being considered, this approach was restricted to models examining only linear terms. As a result, this approach failed to produce a significant improvement over stepwise deletion, and so was not considered further.

Another avenue that was considered for adding waders to the sensitivity map was to include Sites of Special Scientific Interest (SSSIs) that were designated for either Golden Plover or Dunlin. However, information on the SSSI network is not currently available in a suitable format to distinguish SSSIs where Golden Plover and/or Dunlin were present in important numbers as part of a qualifying assemblage of breeding birds, although it is likely to become available in the future

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This appendix was written as part of a project to create a sensitivity map to aid the location of onshore wind farms in Scotland with respect to a suite of sensitive bird species. Thanks to Rowena Langston and James Pearce-Higgins for comments on an earlier draft of this appendix.

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Table A2.1. Sources of Golden Plover abundance data used in the model

Data Source	n	Years collected	Reference
Repeat Upland Bird Survey (RUBS)	1308	1998, 2000, 2002	Sim *et al.*, 2005
Uist wader surveys	152	2000	Jackson *et al.*, 2004
SNH survey of selected SSSIs in the Caithness and Sutherlands Peatlands SPA	138	2004	SNH, 2004
Breeding Bird Survey	123	1994-2003	See Raven *et al.*, 2005 for methods
Grouse Moor Wader Survey	104	1995	Tharme *et al.*, 2001
Moray Moor Wader Survey	33	1997	Francis *et al.*, 1999
Upland Grazing Study	32	1999, 2000, 2003	Pearce-Higgins & Grant, 2006

Table A2.2. Land cover categories from LCM2000 included in models to predict Golden Plover abundance

Term	Land cover categories included
Bog	Bog
Open dwarf shrub	Open dwarf shrub
Dense dwarf shrub	Dense dwarf shrub
Montane habitats	Montane habitats
Woodland	Broad-leaved / mixed woodland + coniferous woodland
Avoided habitats	Sea/estuary + water (inland) + littoral rock + littoral sediment + salt marsh + supra-littoral rock + supra-littoral sediment + set-aside + urban + suburban
Grassland	Neutral grassland + calcareous grassland + acid grassland
Improved grass and arable	Improved grassland + arable horticultural + arable cereals + arable non-rotation

Table A2.3. GLM to predict Golden Plover abundance, including Natural Heritage Zone

Term	d. f.	Chi-Square	p-value
Minimum adequate model:			
Natural Heritage Zone	3	38.07	<0.0001
Bog	1	0.00	0.960
Open dwarf shrub	1	1.25	0.263
Montane habitats	1	5.43	0.020
Woodland	1	1.33	0.249
Avoided habitats	1	0.09	0.758
Grassland	1	1.88	0.170
Forest within 1km	1	0.35	0.551
Forest within 3km	1	6.72	0.010
Mean elevation	1	21.41	<0.0001
Slope	1	20.38	<0.0001
x	1	7.58	0.006
y	1	5.70	0.017
x*x	1	4.10	0.043
x*y	1	24.24	<0.0001
y*y	1	4.65	0.031
Bog * NHZ	3	17.75	0.0005
Open dwarf*NHZ	3	28.22	<0.0001
Woodland*NHZ	2	6.98	0.031
Avoided habitats*NHZ	2	11.03	0.004
Grassland*NHZ	3	10.76	<0.0001
Forest within 1km*NHZ	3	11.25	0.011
Mean elevation*NHZ	3	59.51	<0.0001
x*NHZ	3	49.48	<0.0001
y*NHZ	3	33.56	<0.0001
Excluded terms:			
Improved grass and arable	1	0.00	0.964
Dense dwarf shrub	1	0.09	0.761
Grouse moor management	1	3.80	0.051

Table A2.4. "Critical" densities (after methodology of JNCC, 1998) of Golden Plover for SSSI designation, based on data from surveys using methods after Brown & Shepherd (1993), in five regions of Scotland

	Critical density (pairs per km^2)	Source
Shetland	1.40	Bates *et al.*, 1994a
Caithness & Sutherland	2.09	Arnott *et al.*, 1993; Bates *et al.*, 1994b
Lewis	3.75	Bates *et al.*, 1994c
Moray	1.08	Francis *et al.*, 1999
Southern Uplands	1.22	Smith & Green, 2000

Table A2.5. GLM to predict Golden Plover abundance, excluding Natural Heritage Zone

Term	d. f.	Chi-Square	p-value
Minimum adequate model:			
Dense dwarf shrub	1	10.17	0.001
Montane habitats	1	40.91	<0.0001
Woodland	1	12.31	0.0005
Avoided habitats	1	13.14	0.0003
Grassland	1	35.58	<0.0001
Grouse moor management	1	8.33	0.004
Mean elevation	1	12.68	0.0004
Slope	1	8.63	0.003
x	1	24.05	<0.0001
y	1	42.12	<0.0001
x*y	1	26.65	<0.0001
Elevation*elevation	1	41.60	<0.0001
Excluded terms:			
Bog	1	1.68	0.195
Improved grass and arable	1	0.38	0.535
Open dwarf shrub	1	0.96	0.327
Forest within 1km	1	0.28	0.595
Forest within 3km	1	2.12	0.146

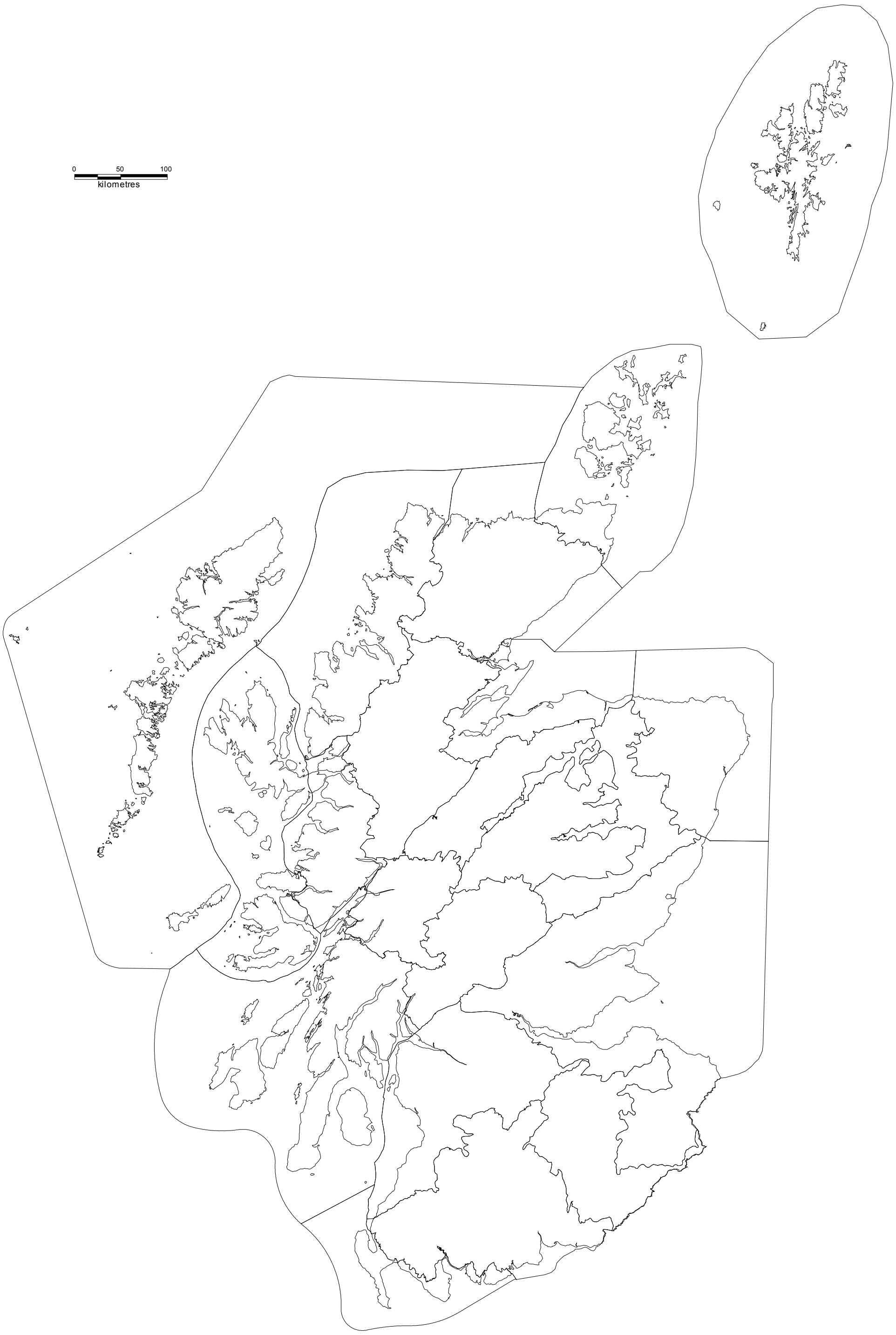


Figure A2.1. Biogeographic zones of Scotland, termed Natural Heritage Zones (NHZs), developed by Scottish Natural Heritage (SNH, 1998).

Appendix 3. Wind farm sensitivity map-building instructions for RSPB staff

The following instructions explain how to create the sensitivity map, based on sensitive bird species, to provide guidance for locating wind farms (see main report). Where a table or column name is being used as an example and should be replaced by the appropriate table or column name, it is written in this font. File names cited in this appendix may have a date as a suffix (e.g. a table named Tetrads may actually be named Tetrads190606). Terms in *italics* refer to commands from the drop-down menu. Command language is written in this colour. The instructions have been written with the aim of explaining how the map was created from scratch from the species' data, but it is intended they could also be used for future updates of the map, for example to add a new species or modify sensitivity ratings. The map is based on sensitive data, and so access to the folder containing species' data is restricted to a few named individuals.

1 Creating individual species' layers

1.1 Individual species' layers were created from the data tables listed in Appendix 3, Table A3.1. Data for each species are contained in the directory: S:\ConSci\Secure\JennyB\Data\Birds*InsertSpeciesName*Data.

1.2 For full explanation of these files see Metadata files.

2 Creating buffers

2.1 These data were then buffered for each species, by distances set out in Table A3.2 (reproduced here from the main report, Table 2). This was done as follows:

2.2 Open the relevant data table in MapInfo (*File, Open Table*). This example uses the Bean Goose data which is in a table called BG.

2.3 Right click on the map window and select *Layer Control.*

2.4 If more than one table is open: left click to select the relevant table, and then move it up the list of tables by clicking on *Up* until it is at the top of the list (just below *Cosmetic Layer*).

2.5 Left click in the second in the row of boxes for the species table to make it editable (it should now look like Box 1).

2.6 Make sure that only the far left box in the *Cosmetic Layer* row contains a tick (as above).

2.7 Click *OK.*

2.8 On the drop-down menu go to '*Query, Select all from* BG'.

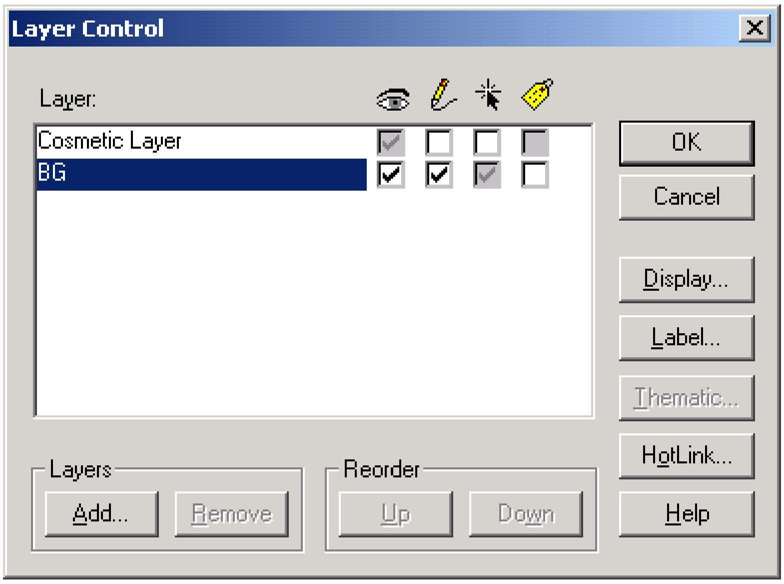
2.9 Right click on the map window and select *Layer Control.*

2.10 Left click in the second in the row of boxes in the *Cosmetic Layer* row, to make it editable. Click *OK.*

2.11 Go to '*Objects, Buffer*' on the drop-down menu. (N.B. Be careful not to do anything else or click elsewhere on the screen in between these two stages, as you may accidentally de-select the Bean Goose data).

2.12 Under *Value,* type in the appropriate distance (see Table A3.2), checking the correct units are selected. Under *Smoothness* type 66, and select O*ne buffer for each object* (as in Box 2).

Box 1

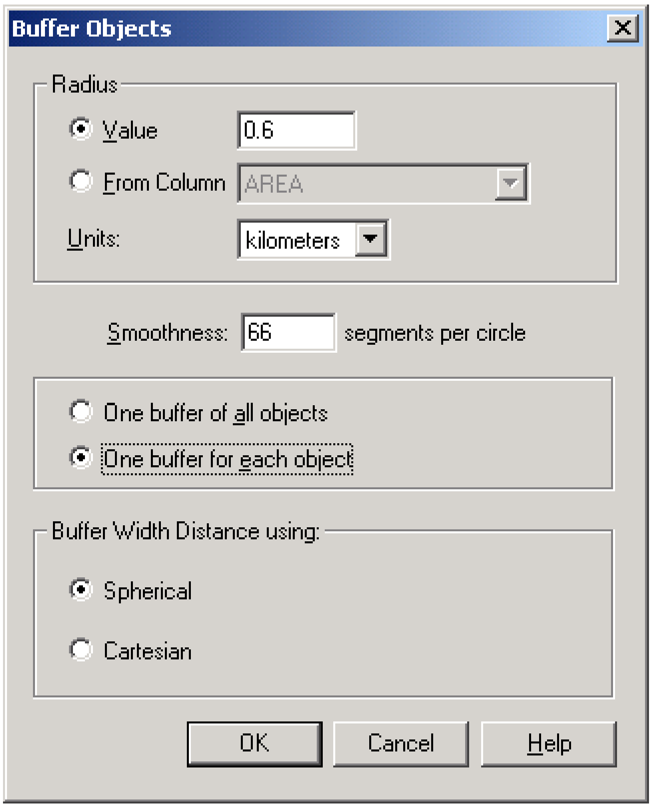


Layer Control

Layer:

Cosmetic Layer	☑	☐	☐	☐
BG	☑	☑	☑	☐

Box 2



Buffer Objects

Radius

● Value 0.6

○ From Column AREA

Units: kilometers

Smoothness: 66 segments per circle

○ One buffer of all objects

● One buffer for each object

Buffer Width Distance using:

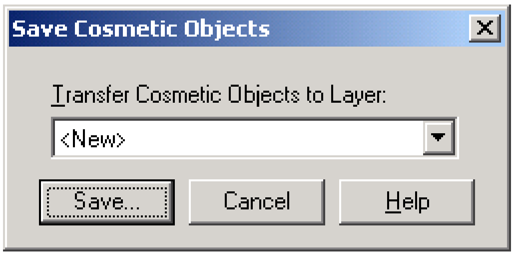
● Spherical

○ Cartesian

2.13 Click *OK*.

2.14 Save the buffers as a separate table by going to '*Map, Save Cosmetic Objects*' under the drop-down menu. The following box appears:

Box 3



2.15 Click on *Save…*and type in the table name in the appropriate directory. Here they were saved as BGBuffers in the directory S:\ConSci\Secure\JennyB\Data\Birds\BeanGooseData.

3 Creating the sensitivity map table

3.1 A 1km square grid of Scotland was created using tables tblScotland and tblgrid1km in the directory C:\Griffin\LocalTableDir (which is on all Merlin PCs) as follows:

3.2 Go to '*Options, Show MapBasic Window*' if it is not already open. This opens up MapInfo's command line window. Minimise windows such that this window and the map windows can be viewed at the same time.

3.3 Type the following (N.B. Press return after entering command lines):

```
Select * from tblgrid1km, tblScotland where tblgrid1km.obj intersects tblScotland.obj
```

3.4 This creates a table of all 1km squares within, or intersecting with the outline of, Scotland.

3.5 Save this table using '*File, Save Copy As*' under the drop-down menu. The file was named ScotlandMap and saved in the directory S:\ConSci\Secure\JennyB\Data\Birds\SensitivityMaps.

3.6 Open this table (it is not opened automatically) using the '*File, Open Table*' option from the drop-down menu.

3.7 Add columns for each species on the sensitivity map, plus one entitled 'Sensitivity' for the sensitivity ratings as follows:

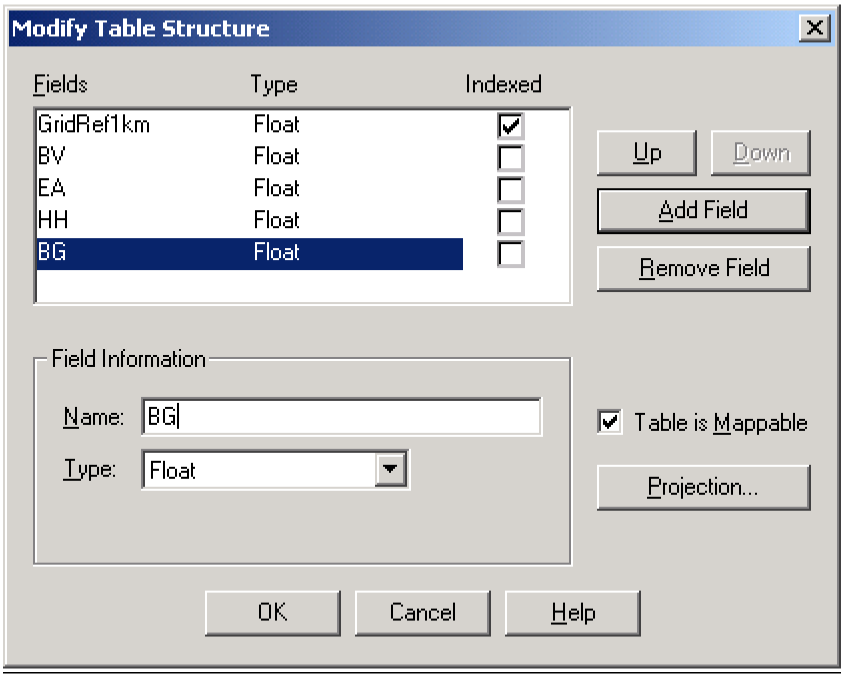
3.8 Select '*Table, Maintenance, Table Structure*' from the drop-down menus and then select ScotlandMap.

3.9 Add columns by clicking on *Add Field*, typing in the appropriate name, and selecting *Type* Float (see Box 4).

3.10 Click *OK* (multiple fields can be added before clicking *OK*).

3.11 The warning shown in Box 5 will appear. Click *OK*.

Box 4



Fields	Type	Indexed
GridRef1km	Float	☑
BV	Float	☐
EA	Float	☐
HH	Float	☐
BG	Float	☐

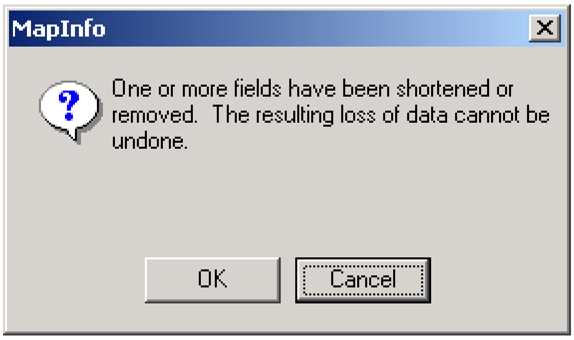
Field Information

Name: BG

Type: Float

☑ Table is Mappable

Box 5



One or more fields have been shortened or removed. The resulting loss of data cannot be undone.

4 Creating species' layers within the ScotlandMap table

4.1 Individual species' columns in the ScotlandMap table are created as follows:

4.2 Open the buffer table for that species (e.g. BGBuffers).

4.3 Open the MapBasic window if it is not already open (*'Options, Show MapBasic Window'*).

4.4 Type the following:

```
Select * from ScotlandMap, BGBuffers where ScotlandMap.obj intersects BGBuffers.obj
```

4.5 This selects all 1km squares intersecting the buffer. For species where 1km squares within the buffer were selected, replace the word 'intersects' with 'within' (see Table A3.1 for which these are).

4.6 Then assign the appropriate sensitivity rating to these squares in the relevant column (Table A3.2 for sensitivity ratings). In this example, we wish to assign a high sensitivity rating to the squares that we have selected (those intersecting with the Bean Goose buffer), so type:

```
Update selection set BG=2
```

4.7 N.B. Be careful not to press anything in between typing these two sentences as you may accidentally de-select the selected squares.

4.8 **NOTE:** Numbers used for different sensitivity ratings in the ScotlandMap table were as follows:

High sensitivity = 2;
Medium sensitivity = 1;
Low/unknown sensitivity = 0.

These differ from the numbers used to represent different sensitivities in the report and Table A3.2, where high sensitivity is '1', medium sensitivity '2' and low/unknown sensitivity is '3'. Different ratings have been used in the table to make it easier to calculate cumulative sensitivities.

4.9 When sending data to other users, these have been changed to the 1, 2 and 3 ratings mentioned above, to match with what is written in the report. This can be done by typing the following into the MapBasic window (press return after each line):

```
Select * from ScotlandMap where Sensitivity=0
Update selection set Sensitivity=3
Select * from ScotlandMap where Sensitivity=1
Update selection set Sensitivity=4
Select * from ScotlandMap where Sensitivity=2
Update selection set Sensitivity=1
Select * from ScotlandMap where Sensitivity=4
Update selection set Sensitivity=2
```

4.10 FOR SPECIES WITH CONCENTRIC RING BUFFERS (Golden Eagle, Nightjar):

- Create two buffer tables as in section two: one with the larger buffer in (e.g. EABuffersMed), and one with the smaller buffer (e.g. EABuffersHigh).
- Select the squares within the outer buffer, in a similar way to that described above for Bean Goose, and assign these a medium sensitivity rating. E.g. For Golden Eagle type:

```
Select * from ScotlandMap, EABuffersMed where ScotlandMap.obj within EABuffersMed.obj
Update selection set Sensitivity=1
```

- THEN, select the squares within the inner buffer, and assign these a high sensitivity rating. E.g. For Golden Eagle, type:

```
Select * from ScotlandMap, EABuffersHigh where ScotlandMap.obj within EABuffersHigh.obj
Update selection set Sensitivity=2
```

- This will over-write the medium sensitivity rating in these squares. MapInfo does have an option to create concentric ring buffers (i.e. to create a donut shaped outer buffer). However, queries on concentric ring buffers take a lot longer than those on circular buffers.

4.11 FOR SPECIES WITH MEDIUM SENSITIVITY RATINGS FOR PREVIOUSLY OCCUPIED AREAS (Golden Eagle, Hen Harrier, Slavonian Grebe):

- Select squares within the previously occupied ranges first (by buffering the older data and selecting squares within these) and assign these a medium sensitivity rating (`Update selection set Sensitivity=1`).
- THEN select squares within the currently occupied ranges (by buffering the new data and selecting squares within these) and assign these a high sensitivity rating (`Update selection set Sensitivity=2`).
- It is important that the ratings are assigned in this order so that if a square forms part of both a previously occupied and a currently occupied range the high sensitivity rating overwrites the medium sensitivity rating.

5 Creating the 'Sensitivity' column in the ScotlandMap table

5.1 Once the single species' columns, plus the SPA column, have all been filled in, update the sensitivity column to contain the highest level of sensitivity rating for any species.

5.2 Go to '*Table, Update Column*'.

5.3 Under *Table to Update* select ScotlandMap, under *Column to Update* select Sensitivity, and under *Get Value From Table* select ScotlandMap (as below).

Box 6.

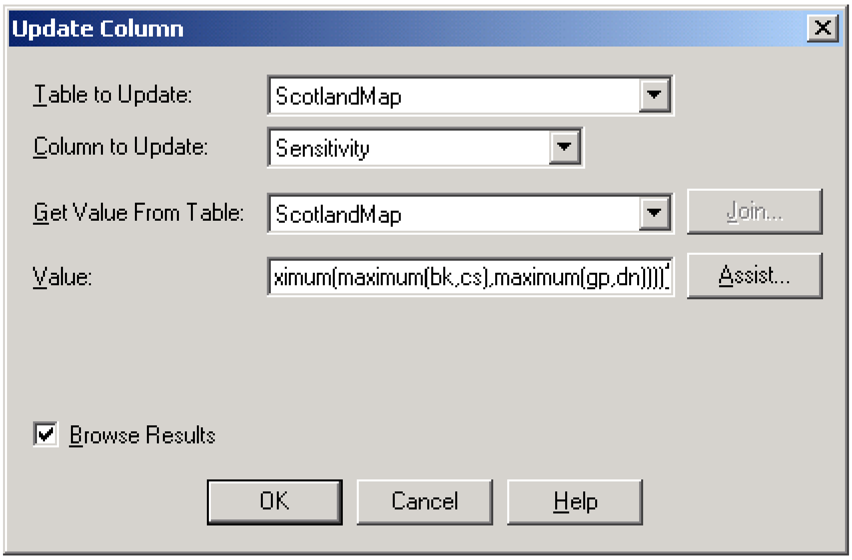


Table to Update: ScotlandMap

Column to Update: Sensitivity

Get Value From Table: ScotlandMap

Value: ximum(maximum(bk,cs),maximum(gp,dn))))

☑ Browse Results

5.4 To calculate the maximum rating, input the following equation in the *Value* field, ensuring there are no spaces (N.B. MapInfo's *Maximum* command only works for two species at a time, so it is not sufficient to type 'maximum(species 1, species 2, species 3'… etc)):

Maximum(Maximum(Maximum(CF,AC),Maximum(Maximum(Maximum(Maximum(BV,EA),Maximum(HH,WTE)),maximum(Maximum(RK,BG),maximum(CP,CE))),maximum(maximum(maximum(NJ,SPA),maximum(RH,SZ)),maximum (maximum(BK,CS),maximum (GP,DN))))),PE)

Equations for calculating the maximum sensitivity rating for other numbers of species are stored in: S:\ConSci\Secure\JennyB\Data\Maximum.doc.

See note, point 4.8 for explanation of sensitivity ratings, and why a maximum rating is used (e.g. why a high sensitivity rating is indicated by 2 in this table, but 1 in the main report).

6 Map Presentation: 1km square map

6.1 The map is presented as a 'thematic map' so that squares with different sensitivity ratings appear as different colours.

6.2 The format of the map is saved in a MapInfo Workspace. To open this go to '*File, Open Workspace*', open the workspace SensitivityMap030506 in the directory S:\ConSci\Secure\JennyB\Data\Birds\SensitivityMaps.

6.3 Any updates made to the ScotlandMap table will automatically change the appearance of the map in this workspace.

6.4 If changes are made to the ScotlandMap table whilst this workspace is open, save the ScotlandMap table ('*File, Save Table*'), close the workspace without saving it (the workspace), and then re-open the workspace. This way, once re-opened the workspace will be based on the updated table, but the overall layout of the workspace will be unchanged.

6.5 To save the map as a picture, left click on the Layout Window and go to '*File, Save Window As*' before saving it as a bitmap or emf, which can be inserted into a word or MS PowerPoint file.

7 Map Presentaion: Creating the tetrad scale map

7.1 The tetrad scale map is not linked to the ScotlandMap table, and must be recreated from the ScotlandMap table as follows each time it is updated (this is slightly complex):

7.2 Open the ScotlandMap table (*File, Open Table*).

7.3 Open the table Grid2kmScot in S:\ConSci\Secure\JennyB\Data\Other. This is a map of all the tetrads either within, or partly within, Scotland.

7.4 Save a copy of the table Grid2kmScot (*File, Save Copy As*) and call it Tetrads.

7.5 Open the Tetrads table (*File, Open Table*).

7.6 Add a column to this table called SensHi (*Table, Maintenance, Table Structure,* then click *Add Field*, name the field SensHi, select *Type=Float*. This is similar to is the procedure shown in Box 4).

7.7 Repeat to add columns called SensMed and Sensitivity (N.B. All three columns can be added before clicking *OK*).

7.8 Click *OK.*

7.9 The warning shown in Box 5 will appear.

7.10 Click *OK.*

7.11 Now create two new tables from the ScotlandMap table, one containing all the high sensitivity squares, and one containing all the medium sensitivity squares. This is done as follows:

7.12 Open the MapBasic Window if it is not already (*Options, Show MapBasic Window*).

7.13 To create a table containing all the high sensitivity squares, type:

```
Select * from ScotlandMap where Sensitivity=2
Browse * from selection
```

7.14 Then go to *File, Save Table As* and save this table as ScotlandHigh (the current name of the table will be written along the top of it and will be e.g. Query1/ Query2…).

7.15 Now, to create a table containing all the medium sensitivity squares, type:

```
Select * from ScotlandMap where Sensitivity=1
Browse * from selection
```

7.16 Then, go to *File, Save Table As* and save this table as ScotlandMed (again, the current name of the table will be written along the top of it and will be e.g. Query1, Query2…).

7.17 Open the tables ScotlandHigh and ScotlandMed (*File, Open Table*).

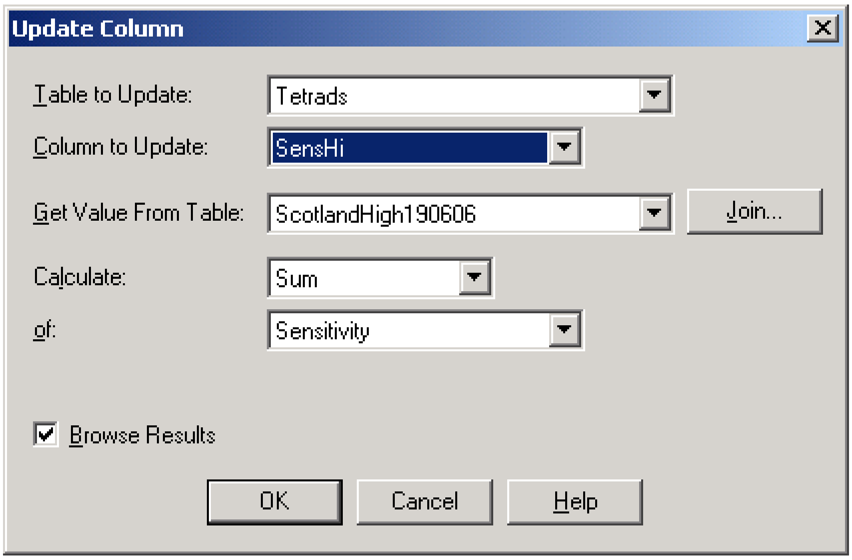
7.18 Input the sum of all high sensitivity 1km squares within a tetrad into the SensHi column in Tetrads as follows:

7.19 Go to *'Table, Update Column'*

7.20 Under *Table to Update* select Tetrads, under *Column to Update* select SensHi under *Get Value from table* select ScotlandHigh

7.21 Under *Calculate* select *Sum*. Under *of* Select Sensitivity (as below). N.B. At this point, the selection under *Column to Update* may have automatically changed to 'Add new temporary column', in which case change it back to SensHi.

Box 7



Update Column

Table to Update: Tetrads

Column to Update: SensHi

Get Value From Table: ScotlandHigh190606

Calculate: Sum

of: Sensitivity

☑ Browse Results

7.22 The column SensHi in the Tetrads table now contains the sum of all high sensitivity ratings within 1km squares in that tetrad. These will be as follows:

Table A3.3

Number of high sensitivity 1km squares within the tetrad	Number in SensHi
0	0
1	2
2	4
3	6
4	8

7.23 Now repeat this so that the SensMed column in the table Tetrads reflects the number of medium sensitivity 1km squares within that tetrad:

7.24 Go to *'Table, Update Column'*

7.25 Under *Table to Update* select Tetrads, under *Column to Update* select SensMed under *Get Value From Table* select ScotlandMed.

7.26 Under *Calculate* select *Sum*. Under *of* Select Sensitivity (this is similar to the information shown in Box 7).

7.27 The column SensMed in the Tetrads table now contains the sum of all medium sensitivity ratings within 1km squares in that tetrad. These will be as follows:

Table A3.4

Number of medium sensitivity 1km squares within the tetrad	Number in SensMed
0	0
1	1
2	2
3	3
4	4

7.28 We now want to update the column to assign a tetrad a sensitivity rating relating to the number of high sensitivity 1km squares within it if it contains any highly sensitive 1km squares, and a sensitivity rating reflecting the number of medium sensitivity squares it contains if it contains no high sensitivity 1km squares. To do this, type the following:

```
Select * from Tetrads where SensMed=1
Update selection set Sensitivity=1
Select * from Tetrads where SensMed=2
Update selection set Sensitivity=2
Select * from Tetrads where SensMed=3
Update selection set Sensitivity=3
Select * from Tetrads where SensMed=4
Update selection set Sensitivity=4
Select * from Tetrads where SensHi=2
Update selection set Sensitivity=10
Select * from Tetrads where SensHi=4
Update selection set Sensitivity=20
Select * from Tetrads where SensHi=6
Update selection set Sensitivity=30
Select * from Tetrads where SensHi=8
Update selection set Sensitivity=40
```

7.29 Save the table Tetrads (*'File, Save Table'*).

7.30 Then, open the workspace TetradMap, which will create a thematic map with the depth of red or yellow of a tetrad dependent upon the ratings above.

Table A3.1 Data used for each species

Species	Data Used	Location of Data	Select Squares within or intersecting with buffers
Red-throated Diver	National Survey 1994, various regional datasets since 1994.	RH1994Survey5km.tab RHAndrewStevenson.tab RHShetland.tab RHNoShetland.tab RHExtraRogerBroad.tab RHB3MerlinNoShetland.tab RHB2MerlinNoShetland.tab RHB1MerlinNoShetland.tab	Intersecting
Black-throated Diver	Surveys from last 10 years, knowledge of commonly used feeding lochs.	BTDLochs2.tab BVFeedingLochs.tab	Intersecting
Slavonian Grebe	Breeding lochs used 1995-2005	SZLochs95to05.tab	Intersecting
	Breeding lochs used 1990-1995	SZLochs90to95.tab	Intersecting
Bean Goose	All fields used on Slammanan Plateau	BG.tab	Intersecting
Common Scoter	National Survey 1995	CSNS95Breeding.tab	Intersecting
Red Kite	All data since re-introduction	RKAllRoosts.tab	Within
	All data since re-introduction	RKAllNests.tab	Within
White-tailed Eagle	All data since re-introduction	WEData.tab	Within
Hen Harrier	National Survey 2004	HH04NS.tab	Within
	National Survey 1998	HHNS98.tab	Within
Golden Eagle	National Survey 2003	EANS03Active.tab EANS03Nests.tab	Within
	National Survey 1992	EANS92.tab	Within
Peregrine Falcon	National Survey 2002	PE2002NS.tab	Within

Species	Data Used	Location of Data	Select Squares within or intersecting with buffers
Black Grouse	Various regional datasets, National Survey 2005	BKActiveLek.tab	Within
Capercaillie	Upper survey strata used for the 2003-04 National Survey. Confirmed breeding/present in 2003, present in 2000	CP1.tab	Intersecting
Corncrake	National Surveys 1993 to 2003	CECentroids.tab	Intersecting
Golden Plover	New Atlas of Breeding Birds 1988-91, mean tetrad counts	In 'BreedingWaderData' folder. GPAtlasTop20Percent.tab	Within
Dunlin	New Atlas of Breeding Birds 1988-91, mean tetrad counts	In 'BreedingWaderData' folder. DNAtlasTop22Percent.tab	Within
Arctic Skua	Seabird 2000	ACover10.tab	Intersecting
Nightjar	National Survey 2004	NJNS04.tab	Within
Chough	National Survey 2002	CFNS02.tab	Within
SPAs	All SPAs	SPAScot210306.tab	Intersecting

Table A3.2 Summary of Sensitivity Criteria for each species

Species	Data Used	Units	Spatial Resolution	Buffer zone	Sensitivity
Red-throated Diver	National Survey 1994, various regional datasets since 1994.	Possible/ probable/ confirmed breeding records	6 figure/ 1km squares	1km	1
Black-throated Diver	Surveys from last 10 years, knowledge of commonly used feeding lochs.	Apparently occupied territories	Loch boundary	1km	1
Slavonian Grebe	Breeding lochs used 1995-2005	Lochs with pair(s) present	Loch boundary	1km	1
	Breeding lochs used 1990-1995	Lochs with pair(s) present	Loch boundary	1km	2
Bean Goose	All fields used on Slammanan Plateau	Fields	Field boundary	0.6km	1
Common Scoter	National Survey 1995	Possible/ probable/ confirmed breeding records	6 figure	1km	1
Red Kite	All data since re-introduction	Nest locations	6 figure/ 1km squares	3km	1
	All data since re-introduction	Roost sites	6 figure/ 1km squares	5km	1
White-tailed Eagle	All data since re-introduction	All territory centres plus known nest locations	1km square	5km	1
Hen Harrier	National Survey 2004	Sightings/ nest locations	6 figure/ 1km squares	2km	1
	National Survey 1998	Sightings/nest locations	6 figure/ 1km squares	2km	2
Golden Eagle	National Survey 2003	Nest locations/ territories	6 figure/ 1km squares	2.5km	1
	National Survey 2003	Nest locations/ territories	6 figure/ 1km squares	2.5km to 6km	2
	National Survey 1992	Nest locations/ territories	6 figure/ 1km squares	6km	2

Species	Data Used	Units	Spatial Resolution	Buffer zone	Sensitivity
Peregrine Falcon	National Survey 2002	Nest locations/ territories	6 figure/ 1km squares	2km	2
Black Grouse	Various regional datasets, National Survey 2005	Lek sites	6 figure/ 1km squares	1.5km	2
Capercaillie	Upper survey strata used for the 2003-04 National Survey. Confirmed breeding/present in 2003, present in 2000	Forest blocks over 0.5ha containing Capercaillie since 2000	Boundaries of forest blocks over 0.5ha	0km	1
Corncrake	National Surveys 1993 to 2003	Singing males	6 figure	0.85km	1
Golden Plover	New Atlas of Breeding Birds 1988-91, mean tetrad counts	Adults	10km squares with top 20% of mean counts	0km	2
Dunlin	New Atlas of Breeding Birds 1988-91, mean tetrad counts	Adults	10km squares with top 20% of mean counts	0km	2
Arctic Skua	Seabird 2000	Colonies of 10 or more birds	1km square	0km	1
Nightjar	National Survey 2004	Churring males	1km square	1km	1
	National Survey 2004	Churring males	1km square	1km to 2.5km	2
Chough	National Survey 2002	Nest locations	1km square	1km	1
SPAs	All SPAs	SPAs	SPA boundary	0km	1